Integrating erosion risk into forest management in Catalonia, Spain

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Academic dissertation

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ABSTRACT

Soil erosion processes are common in Mediterranean areas due to its climatic conditions, abrupt topography, and its long history of human influence and intensive land management. Forests have a fundamental protective role in combatting soil erosion and land degradation. This research focused on i) assessing soil erosion on forest lands linked to forest variables, and ii) integrating erosion risk into forest management and planning. In more detail, the first goal focused on the relation between erosion occurrence and possible influencing factors such as forest structure and composition, site and land-use characteristics, and, based on this, integrating erosion protection as an objective into multi-functional forest management. The data source was the Spanish National Forest Inventory, which provided erosion records as well as traditional tree and plot measurements. The modelling methods used included classification trees and binary logistic regression analysis. Models were developed for the stand and landscape levels to predict the probability of erosion. Erosion risk was also studied spatially by producing maps based on the predictions of the models. Finally, the models were integrated into a simulation optimisation system in order to assess management alternatives.

The results show that, at the stand level, a higher probability of surface erosion occurrence was strongly related to the forest type and stand structure. For example, sparse stands on slopes in semiarid areas and dense stands with large trees in pure *Abies alba* and *Facus sylvatica* forest had higher erosion probability. At the landscape level, a low stand basal area, *andisol* and *cambisol* soil types, large size and steep slope of the drainage area, the presence of unpaved roads and increased urban land use resulted in a higher probability of gully erosion. Based on the developed models, erosion protection was used as the management objective in multi-functional forest planning at the stand level, together with timber production and structural diversity. The results demonstrated evident trade-offs between these ecosystem services, depending on the steepness of the slope. The results of this work can help to identify forest areas vulnerable to soil erosion. It provides tools to integrate erosion risk into forest planning, thus enhancing the use of forests in erosion protection in sensitive Mediterranean areas.

ACKNOWLEDGEMENTS

LIST OF ORIGINAL ARTICLES

This thesis is based on the following three articles, which are referred to the text with Roman numerals **I-III**. Articles are reprinted with the kind permission of the publishers.

I   **Selkimäki, M**., González-Olabarria, J.R., Pukkala, T. (2012). Site and stand characteristics related to surface erosion occurrence in forests of Catalonia (Spain). European Journal of Forest Research 131: 727–738.
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II   **Selkimäki, M**., González-Olabarria, J.R. (2016). Assessing gully erosion occurrence in forest lands in Catalonia (Spain). Land Degradation & Development 28: 616–627.
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III   **Selkimäki, M**., González-Olabarria, J.R., Trasobares, A., Pukkala, T. (2020). Trade-offs between economic profitability, erosion risk mitigation and biodiversity in the management of uneven-aged *Abies alb*a Mill. stands. Annals of Forest Science 77:12.
https://doi.org/10.1007/s13595-019-0914-z

Author's contribution

Mari Selkimäki was responsible for data preparation and analyses, compiling and writing of the thesis and manuscripts. Co-authors participated in the planning of the studies, discussion of the results and commenting the papers. In paper **III** the integration of models into simulation-optimization software was done by T. Pukkala and the growth models of *Abies alba* by A. Trasobares.

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ABBREVIATIONS AND DEFINITIONS

Abbreviations

NFI	National Forest Inventory
CART	Classification and Regression Trees
DEM	Digital Elevation Model
ES	Ecosystem Services
EP	Erosion Protection
NPV	Net Present Value
ROC	Receiving Operating Characteristics curve
AUC	Area Under the Curve

Definitions

Surface erosion	Tree roots are exposed, signs of water-accumulated materials above obstacles and an abundance of exposed stones.
Rills	Removal of soil from parallel small streams not deeper than 30 cm.
Gully	Removal of soil along a drainage channel created by running water into topographic depressions, deeper than 30 cm.
Ecosystem services	Benefits for humans provided by ecosystems. They include provision services such as food, water, timber; regulating services such as climate regulation, erosion control; and cultural services such as recreation, spiritual and religious values.
Trade-off	An increment in one ecosystem service decreases the amount of another service.

INTRODUCTION

Mediterranean forests and soil erosion

Mediterranean forests provide several ecosystem services (ES) such as timber, cork, resin, mushrooms, biodiversity, carbon storage, soil protection, watershed management and cultural and recreational values (e.g. Merlo and Croitoru 2005; Palahí et al. 2008; EME 2011). Most of the forests across the Mediterranean region are heterogeneous due to their long history of human influence, including intensive land management. Large variations in the topography and rainfall regime have also affected vegetation characteristics, resulting in a diversity of landscapes and biodiversity hotspots for both animal and plant species (Blanco et al. 1997; Palahí et al. 2008; FAO 2018). The environmental conditions make the forests vulnerable to disturbances due to irregular rainfall, high temperatures, periods of drought and frequent forest fires. These phenomena can reduce the vitality of plants and lead to the loss of vegetation cover and biodiversity, which can increase land degradation.

The climatic conditions, biogeography, soil characteristics and long history of human influence through intensive land management favour the development of soil erosion processes in the Mediterranean region (Palahí et al. 2008; García-Ruiz et al. 2013). The most common soil erosion is caused by water. The process has three states: detachment, transport and deposition (Morgan 2005). In the process of water erosion, detachment is induced by rainfall and displacement of soil particles by raindrops dropping from clouds or tree canopies, while transport of soil particles is caused by overland flow, which can also cause soil particles to detach. Both detachment and transport processes need a certain amount of energy to cause erosion (velocity and kinetic energy). The energy of rainfall and water run-off depends on the intensity of the rainfall, size of the raindrops and steepness and length of the slope (Blanco and Lal 2008). Soil characteristics affect water infiltration rates and surface run-off. They define how much energy is needed to detach and transport soil particles (Morgan 2005). When transport energy reduces and is no longer sufficient to carry soil particles, deposition occurs, causing sedimentation.

Vegetation and litter cover usually protect the soil from rain splash erosion, depending on coverage, vegetation type and plants' morphological structure (Boer and Puigdefábregas 2005; Bochet et al. 2006). Raindrops falling from high tree canopies tend to be larger and have higher terminal velocities compared to free rainfall, and therefore can cause more erosion under tall tree canopies if the underground vegetation is sparse compared to dense shrub vegetation cover (Brandt 1987; Calder 2001; Nanko et al. 2004; Kang et al. 2008). The forms of water erosion at the local scale include surface (also called sheet) erosion caused by raindrops and surface flow, and rill erosion, defined as small shallow channels formed by concentrated overland flow shallow enough to be removed by tillage operations. Large-scale water erosion processes include gullies, deep channels formed in topographic depressions by concentrated overland flow that cannot be removed by tillage operation (Morgan 2005). Gullying is an important land degradation process which reduces site productivity and limits land uses. Eventually, it can lead to actively developing badlands, especially in semiarid areas (Lesschen et al. 2007; Gómez Gutiérrez et al. 2009; García-Ruiz et al. 2013).

Land degradation by soil erosion processes does not only cause on-site effects such as deterioration of ecosystems due to reduction in soil quality and water availability but also economic losses due to the decreasing productivity of agricultural and forest land. The off-site effects of sediment siltation on rivers can reduce the water quality and the capacity of

reservoirs. This can potentially increase the number and intensity of floods and cause water pollution and gaseous emissions such as carbon dioxide and methane (Poesen et al 1996; Blanco and Lal 2008). Soil erosion has been estimated to have caused an economic loss of 280 million euro per year during the 1990s in Spain (ICONA 1991). In Italy, the annual cost is estimated to be around 619 million euro (Panagos et al. 2018). The importance of soil erosion and the protective function of forest lands have been reflected at the EU level, especially in the Mediterranean countries (Borelli et al. 2016; FAO 2018). For example, in Spain, the importance of preventing soil erosion and land degradation has been recognised and is reflected in forest policy (DGCN 2002), which emphasises that the sustainable use and management of forest should combat soil loss and desertification and at the same time preserve biodiversity. This led to massive reforestation programmes during 1945-1986 when approximately 3.8 million hectares were reforested (Pausas et al. 2004a). According to the third Spanish National Forest Inventory (DGCN 2005), 10.5% of the inventory plots suffered from different levels of soil erosion in the region of Catalonia (NE Spain). This shows that erosion on forest land is an important issue which must be considered in forest management and planning.

Erosion risk in forest lands

Soil protection is one of the ES benefits provided by the forest. Soil erosion is a natural part of soil development, but it can be accelerated by human activities (Morgan 2005; Blanco and Lal 2008). Well-known factors influencing erosion risk on forest lands include deforestation and harvesting activities (Croke et al. 2001; Hartanto et al. 2003; Kolka and Smidt 2004) due to machinery operations, skid tracks and the building of forest roads (Elliott et al. 1999; Croke and Mockler 2001; Madej et al. 2006; Wade et al. 2012) to access harvesting sites. The highest soil losses are typically caused by those harvesting operations that include site preparation, such as ploughing (Edeso et al. 1999), which disturb the soil structure. Also the reforestation of old agriculture lands, which includes soil modifications, can increase erosion rates for many years (de Figueiredo et al. 2012). Other significant factors increasing soil erosion risk are forest fires, which destroy the vegetation cover and can alter the soil's physical properties (Thomas et al. 1999; Johansen et al. 2001; Pardini et al. 2004; Hyde et al. 2007; Cannon et al. 2010). In addition, recreation activities and associated infrastructures, such as hiking trails (Selkimäki and Mola-Yudego 2010), ski tracks (Pintaldi et al. 2017) as well as grazing on forest pastures (Strunk 2003) can contribute to soil erosion on forest lands. Erosion processes are not always due to human activities but can also be significant in untreated forest, especially on steep slopes receiving intensive rainfall (Hartanto et al. 2003; Morgan 2005).

A common view is that the denser the forest cover is, the more it protects the soil against erosion processes. Many of the existing erosion models use the percentage of canopy cover as the sole variable describing the effect of the forest structure on erosion. Several models exist to predict potential soil erosion loss, such as USLE (Wischmeier and Smith 1978), its revised version RUSLE (Renard et al. 1997) and WEPP (Flanagan and Livingston 1995; Laflen et al. 1997). These models relate soil erosion on forest lands to vegetation cover factors, typically the percentage of canopy cover. However, this does not provide a sufficient description of the forest structural variables. Therefore, alternative factors have been suggested, e.g. the amount of bare soil, fine roots, the organic matter content of the soil and combinations of tree height and canopy cover. These variables have improved the model

predictions (Dissmeyer and Foster 1980). Some studies suggest that ordinary growing stock variables such as stand density (Razafindrabe et al. 2010) and tree height (Brandt 1987; Calder 2001; Kang et al. 2008) are associated with soil erosion, and could be used to integrate erosion risk into forest management. The use of ordinary forest stand variables such as basal area, stocking and tree height would make it easy to integrate erosion risk into forest management and planning. This is important, since with proper planning and management, the erosion risk of forested landscapes could be reduced significantly. Previous studies show that under favourable conditions, certain forest types with overly dense stands can actually have higher erosion rates compared with less dense stands, and erosion could be reduced by appropriate thinning operations (Nanko et al. 2004; Razafindrabe et al. 2010).

Risk assessment and forest management

Under climate change scenarios, extreme climatic conditions are expected to become more frequent. In the Mediterranean area, this means long periods of drought and extreme precipitation events with high-intensity rainfall resulting in increasing risks of forest fires and soil erosion (IPCC 2013). To take these disturbances into consideration in forest management, the identification of forest structures vulnerable to different disturbances is needed. Several studies based on national forest inventories (NFI) have shown the usefulness of this data source in the evaluation of the risk of different natural disturbances to forests. NFI provides data over large areas and the inventory is typically repeated at regular time intervals. Integrating risk in forest management planning has been undertaken for fire (e.g. González et al. 2005, Costa Freitas et al. 2017), wind (e.g. Gardiner and Quine 2000; Heinonen et al. 2009), snow (e.g. Díaz-Yáñez et al. 2019) and browsing (e.g. Díaz-Yáñez et al. 2017). Yet, for erosion risk there is a lack of similar models, since erosion variables are seldom included in forest inventories.

In Spain, the state of the soil erosion, together with forest structure and composition, was systematically recorded in the third Spanish forest inventory (NFI3). The erosion observations in the NFI plot location were classified into: surface and rill erosion, V-shape gullies, U-shape gullies and landslides. There are several advantages of making erosion observations in the forest inventory. The visual observation of the erosion state can be connected to site and stand variables based on tree- and plot-level measurements. The NFI provides information for large areas, and since the inventory is repeated regularly, it is possible to follow forest development and the state of erosion.

Traditionally, forest management has focused on timber production (Messier et al. 2015), while non-timber products (e.g. berries, mushrooms, resins) and other uses of forests (e.g. erosion control, recreation, water regulation) have received less attention (Rodríguez et al. 2006; Bennet et al. 2009). Multi-objective forest planning methods have been used in the assessment of different forest ecosystem services. Many of the forest ES are in trade-off, meaning that the production of one service has a negative effect on the provision of another service (MEA 2005). The provision of several forest ES has been evaluated under different forest management options, including, for example, timber production, biodiversity, carbon balance and non-wood products such as mushrooms and berries (e.g. Mönkkönen et al. 2014; Diaz-Balteiro et al. 2017; Lafond et al. 2017; Kurttila et al. 2018). Soil protection as an ES has been defined as forests growing on slopes steeper than 30% (e.g. Roces-Díaz et al. 2018). In other cases, assessment of soil protection ES is based on potential soil loss calculation using the USLE (e.g. García-Nieto et al. 2013) or RUSLE model (Anaya-Romero et al. 2016).

In order to assess the trade-off between timber production and soil protection, the relationship between stand structural variables and soil protection needs to be modelled and estimated for different management scenarios. In this thesis, erosion protection as an ES in *Abies alba* mountain forest was assessed by using forest structural variables that can be controlled in forest management. Therefore, the trade-offs between timber production and erosion protection could be studied (study **III**).

Aims of the thesis

The aim of this thesis is to assess soil erosion on forest lands and integrate the erosion risk into forest management and planning in Catalonia. This thesis focuses on two erosion processes recorded in the Spanish NFI, surface and gully erosion. First, soil erosion risk is assessed at the stand and landscape levels. Erosion occurrence is related to the forest structural variables and land uses in the drainage areas, leading to the development of erosion risk models for the region. Secondly, the erosion risk is integrated into the forest management planning system and used as one objective variable in the optimisation of multi-functional forest management.

The specific objectives are:
1. To study the relationship between forest structure, site and stand variables and surface erosion at the stand level (**I**).
2. To study gully erosion occurrence and its relation to landscape structure and land-use changes at the landscape level (**II**).
3. To develop a multi-objective forest planning approach that integrates soil erosion risk into forest management planning at the stand level, and use the approach to analyse trade-offs between three ecosystem services: economic profitability, erosion risk mitigation and structural diversity (**III**).

The main hypotheses of the research are:

1. There are differences in the probability of erosion occurrence between forest stands depending on the forest structure and tree species composition as well as site characteristics.
2. At the landscape level, local forest variables and environmental factors in larger areas, fragmentation and land-use changes correlate with the occurrence of gully erosion.
3. Increased emphasis on erosion protection and biodiversity reduces the economic profit from timber production.

MATERIAL AND METHODS

Study area

The region of Catalonia is located in the north-eastern part of Spain, covering 32 106 km^2. The region has a Mediterranean climate with large climatic variations due to the influence of the mountainous topography (Gracia et al. 2004). Catalonia can be divided into eight climatic regions: semiarid, dry sub-humid, sub-humid, humid I to IV and very humid (Thornthwaite 1948; Clavero et al. 1997). The mean annual precipitation varies from 300 to 1500 mm (Ninyerola et al. 2000) and the altitude ranges from sea level up to 3000 m above sea level. The Pyrenean mountain area in the north receives the highest amount of precipitation, which decreases towards the southern and less elevated areas. Precipitation also decreases from east to west, from the coastal areas of the Mediterranean Sea to the inland Central Depression (Lleida). The dominant soil types in the region are cambisols, fluvisols, regosols and leptosols (Soil Atlas of Europe 2006). Large differences in topography and climatic conditions and a long history of human influence and recurrent forest fires have influenced the evolution and current distribution of forest types in Catalonia. Forests in the region are dominated by pines such as *Pinus halepensis*, *P. sylvestris*, *P. nigra*, *P. uncinata* and *P. pinea*, and oaks such as *Quercus ilex*, *Q. suber* and *Q. humilis*, followed by *Fagus sylvatica*, *Abies alba*, *Castanea sativa* and *Betula pendula*. Pure forests (defined as forest where at least 80% of the basal area belongs to a single tree species) cover approximately 8040 km^2, of which 5540 km^2 are pure coniferous and 2500 km^2 pure broadleaf forest. The rest of the forest lands are mixed forest, covering almost 12 000 km^2 (Gracia et al. 2004).

The landscape of the region has undergone transformations due to reforestation and abandonment of traditional land-use practices during recent decades. Since the 1950s, there has been a steady abandonment of farmland, especially in mountain areas, leading to an increase in the forest land area. In addition, urban areas have been enlarged due to the combined dynamics of population growth and the movement of people from rural areas to cities (Poyatos et al. 2003; García-Ruiz et al. 2005; López-Moreno et al. 2006).

Data sources

The data on soil erosion and forest stand variables were obtained from the Third Spanish National Forest Inventory NFI3 (DGCN 2005), conducted in Catalonia during 2000 and 2001. The NFI data consist of a systematic sample of permanent plots, distributed on a 1 km x 1 km square grid, with a re-measurement interval of about ten years. The sampling design uses concentric circular plots with a radius depending on the tree diameter at breast height (dbh): 5 m radius is used for trees with dbh between 7.5 and 12.4 cm; 10 m for 12.5–22.4 cm; 15 m for 22.5–42.4 cm; and 25 m for trees with dbh greater than or equal to 42.5 cm. The following data were recorded for each sample tree: species, dbh, height and distance and azimuth from the plot centre. The health state of the trees, damage and observed management operations were also recorded. The shrub coverage was assessed by species within a 5 m radius around the plot centre. In every plot the erosion was visually observed up to 60 m around the plot centre and coded into six categories; *no erosion, surface erosion, rill erosion, V-shape gully erosion, U-shape gully erosion* and *landslides*.

The data on forest fires were obtained from fire perimeter maps generated by the *Department de Medi Ambient I Habitatge and Institut Cartográfic de Catalunya*. The plots were marked as burnt if a fire had occurred in their location between IFN2 and IFN3 (between 1989 and 2000). Data on annual mean precipitation were obtained from digital maps produced by the climatic model of Ninyerola et al. (2000), and the climatic division of the region was obtained from the Climatic Atlas of Catalonia (Clavero et al. 1997). Land-use maps of Catalonia included a vector map of 2005 (MCSC 2005) and a raster map of 1987 (Generalitat de Catalunya 1987). Information on soil types was obtained from the European Soil Database raster with 1 km x 1 km cell size (Van Liedekerke et al. 2006). The locations and types of roads were obtained from the database of the Cartographic Institute of Catalonia (De la Base Topográfica 2007). Data on topography (slope, elevation and aspect) were derived from a digital elevation model (DEM) of 30 m x 30 m resolution (Institut Cartogràfic i Geològic de Catalunya).

Erosion observations in the NFI

The NFI plots were classified according to climatic division (Clavero et al. 1997) into semi-arid and humid. The NFI plots in the humid areas were further classified into three forest types: pure coniferous, pure broadleaf (if a single species represented over 80% of the total basal area of the plot) and mixed forest. Plots without any trees with dbh >7.5 cm were classified as *no trees* plots. Table 1 shows the share of erosion types in the different forest types. Of the 11101 NFI plots, 10.5% had some signs of erosion. The aim of study **I** was to discover site and stand variables that were related to surface erosion. Therefore, only plots with at least one tree larger than 7.5 cm were chosen (resulting in 689 plots with surface erosion and 8872 plots without erosion). The landscape-level study **II** on gully erosion included all the *no erosion* plots, also the *no trees* plots (resulting in 157 plots with gully erosion and 9932 plots without erosion). The map in Figure 1 shows the location of the eroded plots used in studies **I** and **II,** and the location of plots dominated by *Abies alba* in study **III**.

Table 1: Number of forest inventory plots by erosion classes (V and U-shape gully erosion are summed) and the percentage of all erosion types by forest types.

Forest type	No erosion	Surface	Rills	Gullies	Landslide	% erosion
No trees	1060	105	16	20	11	12.5
Semiarid	454	98	18	9	2	21.9
Pure coniferous	3338	185	20	49	27	7.8
Pure broadleaf	1620	156	13	24	32	12.2
Mixed	3460	250	28	55	51	10.0
Total	**9932**	**794**	**95**	**157**	**123**	**10.5**
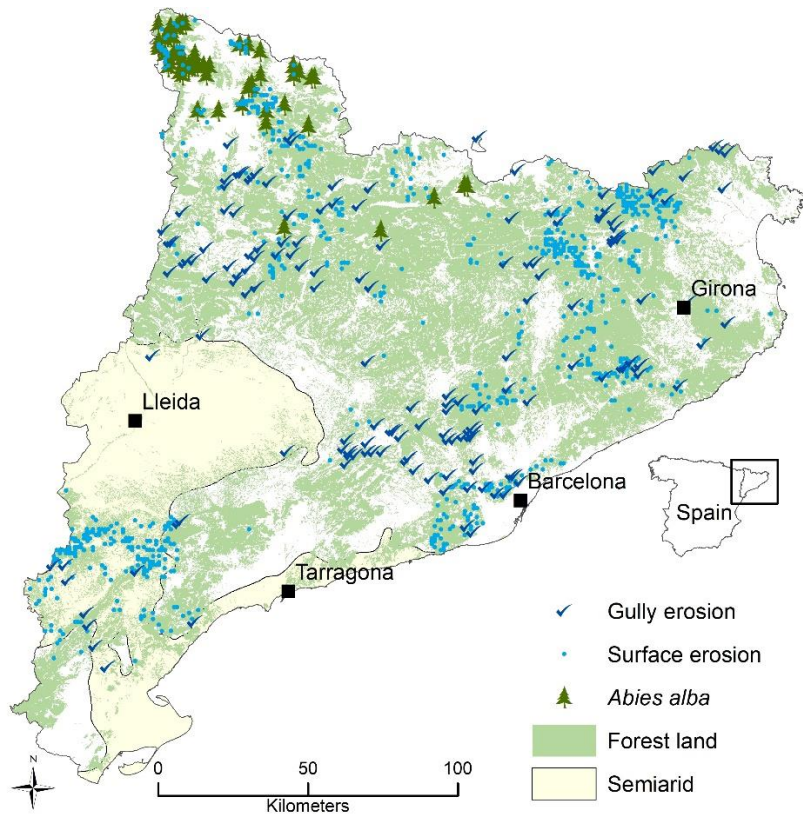
Figure 1: Study region of Catalonia, location of NFI3 surface erosion and gully erosion plots and plots dominated by *Abies alba*.

Modelling surface erosion at the stand level

Classification tree method

The classification and regression trees (CART) method was chosen to explore the complex and non-linear relationships between site and stand variables and surface erosion. As a non-parametric method, CART makes no assumptions on the distribution of the data. It can deal with large datasets having both nominal and continuous variables and it can handle interactions between the predictor variables (Breiman et al. 1984). The method recursively partitions the data into homogenous groups, aiming at node impurity, which in this study means separating all surface erosion plots from non-eroded plots by using the predictor variables and their values as splitting criteria. The splitting procedure evaluates the importance of variables based on their degree of homogeneity by incorporating the prior probabilities and misclassification cost of the variable. The goodness of the splits in the *rpart* package of the R-software used is measured with the Gini index, which maximises the impurity reduction of the split (Therneau and Atkinson 1997).

The CART procedure first grows the classification tree to full size and then prunes it. Pruning the classification tree makes the tree smaller and easier to interpret, and it also improves the probability estimates derived from the tree (Provost and Domingos 2003). In the study, a minimum of 20 plots per node was set as the pruning criterion. The predictor variables used for construction of the classification tree are listed in Table 1 of study **I**.

Instead of using the dominant classes of the classification tree leaves at the end of the branches, the class frequencies were converted into erosion probability estimates. The Laplace correction was used to remove the extreme probabilities caused by the small sample size at the end of the leaves. This is a widely used smoothing method in CART analysis (Provost and Domingos 2003; Ferri et al. 2003; Tóth and Pataki 2007), calculated as follows:

$$P_i = \frac{n_i + 1}{N + C}$$

(1)

where P_i is the smoothed probability and n_i is the number of observed samples of class i at the leaf, N is the total number of samples in the leaf and C is the number of classes at the leaf.

Figure 2 shows the schematic structure of the classification tree. Based on the preliminary analysis, the data were divided by classifying the NFI plots into forest types (level 1). The final classification tree results (level 2) were produced separately for each forest type.
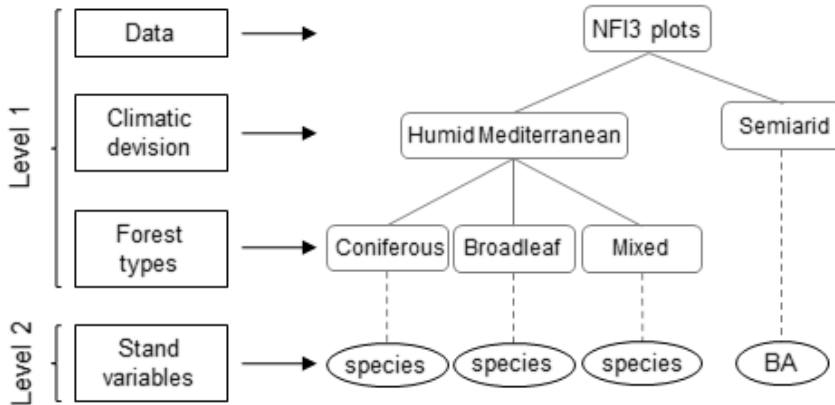

Figure 2: Schematic structure of the two levels of the classification tree.

Binary logistic

The classification tree analyses used in study **I** showed that inventory plots with *Abies alba* and *Fagus sylvatica* as dominant species had a higher occurrence of surface erosion. In study **III**, the aim was to integrate the erosion risk into forest management planning. Pure *Abies alba* forest was chosen to study erosion mitigation as one objective when planning multifunctional forest management. In order to integrate the erosion risk into a simulation optimisation system (*Rodal*, Palahí and Pukkala 2003), the erosion risk was modelled using a logistic function. The same variables were used that were included in the classification tree together with the slope. The model for erosion probability was fitted using binary logistic regression analysis. The model for the probability of erosion was as follows:

$$E_{risk} = \left(1 + e^{-[-12.356+0.04BA+0.301H+0.134dead\%+0.13slope]}\right)^{-1} \tag{2}$$

where *BA* is the basal area (m^2 ha^{-1}), *H* is the mean height on the trees (m), *dead%* is the percentage of standing dead trees and *slope* is the steepness of the slope in degrees obtained from the DEM. The model had good predictive power (Nagelkerke R^2=0.448), its overall percentage of correct classifications being 82.1% with 0.5 cutting value. The Hosmer and Lemeshow test statistic was 0.462 and the area under the receiving operating characteristic (ROC) curve was 0.836. As one of the management objectives in study **III** was to mitigate erosion, the probability of erosion (E_{risk}) was converted into an "erosion protection" (EP) value, calculated as 1-E_{risk}.

Modelling gully erosion at the landscape level

Study **II** analysed gully erosion at the landscape level. Gully erosion consists of erosion channels formed by water flowing into topographic depressions. The assumption of the landscape-level analysis was that the water flow contributing to gully channels comes from upslope. Therefore, the study was based on small-scale drainage areas which were delineated for each NFI plot. In order to cover the total potential water contributing area, the centre of the inventory plot plus 8 points along the 60 m circle from the plot centre (erosion observation were made up to 60 m) were used as pour points. These points were defined as outlets to which the drainage areas were calculated using ArcGIS Hydro Tools 9 and DEM. The final contributing area was aggregated from all nine drainage areas and the flow accumulation (stream line) to the final drainage area was defined (Fig. 3). The following drainage variables were calculated: size of the drainage area, slope of the drainage area along the stream line and length of the stream line.

Drainage area boundaries were used to assess land-use changes and identify which factors had an influence on facilitating or reducing the potential water run-off that could contribute to gully formation. Land-use maps from 1987 and 2005 were overlaid and clipped with drainage area boundaries for the analyses (Fig. 3). Land uses were reclassified into seven classes: water, urban, agriculture, shrub land, forest and bare area (including cliffs, bedrocks and burnt areas). The land-use coverages and their changes over time were calculated for each drainage area and the differences between gullied and non-eroded plot drainage areas were estimated using analysis of variance.

Gully erosion occurrence was modelled using stepwise logistic regression. The potential predictors included plot-level stand variables and drainage area level variables such as land-use change, precipitation, soil type, fire, and presence of roads (full variable list in Table 1, study **II**).
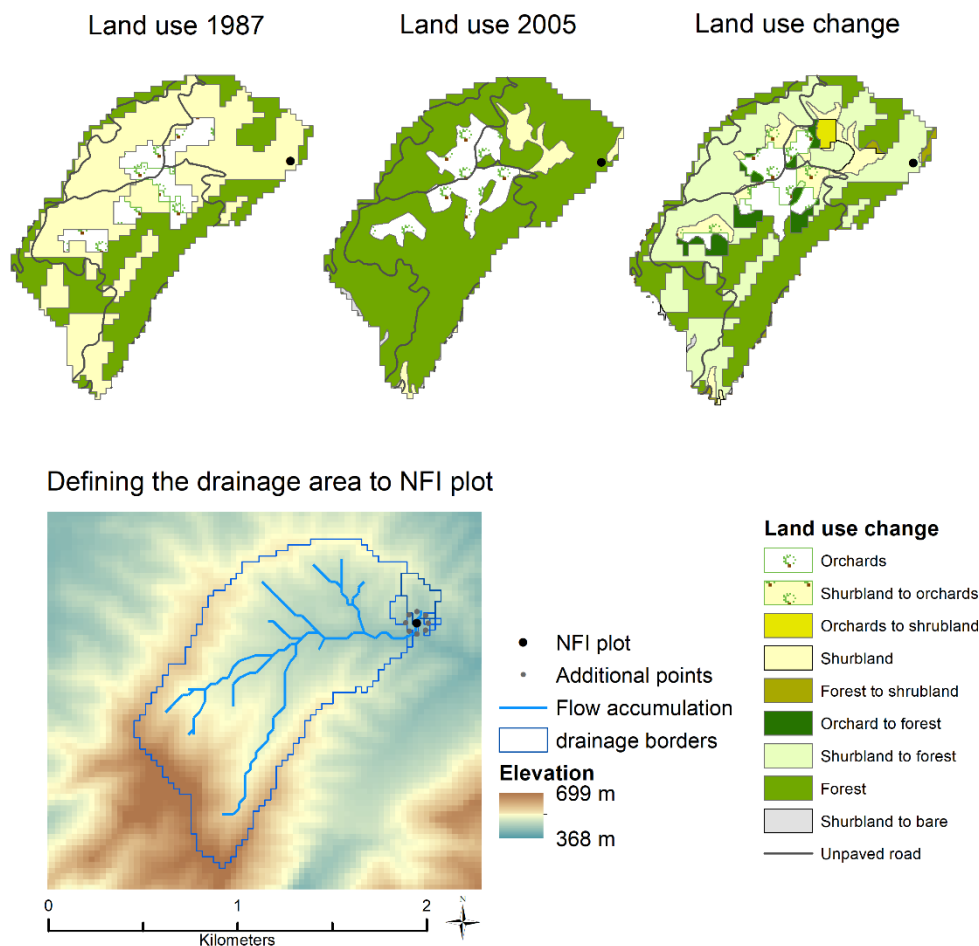

Figure 3: Example of defining the drainage area for an NFI plot. Drainage borders were defined for the NFI plot and to additional points. Separate borders were joined to obtain one drainage area, which was used to clip land uses (1987 and 2005) in order to calculate land-use change in the drainage area.

Integrating the erosion risk into stand-level optimisation

In study **III**, the aim was to optimise the structure and management of uneven-aged *Abies alba* stands by taking into account three ecosystem services provided by the forest: economic profit from timber, erosion protection and biodiversity. For this purpose, the surface erosion risk (Equation 2), once transformed into an erosion protection index, was integrated into the Rodal forest stand simulation optimisation software (Palahí and Pukkala 2003). The management objectives were to 1) gain revenues from timber measured by net present value (NPV), 2) reduce the risk of surface erosion (maximising erosion protection EP), and 3) maintain structural diversity by keeping a minimum of 10 trees/ha of dbh >60 cm. The stand dynamics were simulated and management was optimised for three slopes (10, 45 and 80%) using three different cutting cycles (10, 30 and 50 years) on each slope.

The simulation and optimisation system, once adapted for the specific requirement of the study, was used to find the optimal diameter distribution of the stand. The system optimised the values of four decision variables, maximising the revenues from timber and, at the same time, maximising erosion protection (Fig. 4). Diameter distribution was described using the two-parameter Weibull function (Bare and Opalach 1988). Four optimised decision variables together defined the post-cutting diameter distribution of the trees (total number of trees per hectare, Weibull parameters b and c, maximum retained diameter). Harvesting was conducted in such a way that the original diameter distribution was returned, in order to maintain the uneven-aged stand structure. Penalty functions were added to the objective function to guarantee the sustainability of the diameter distribution and to force a sufficient number of large trees to the solution to maintain high structural diversity. Hooke and Jeeves' (1961) direct search method was used as the optimisation algorithm. It is commonly used in stand management optimisation problems (e.g. Haight and Monserud 1990; Rautiainen et al. 2000; Trasobares and Pukkala 2004; González et al. 2005; Pasalodos-Tato and Pukkala 2007; González-Olabarria et al. 2008).
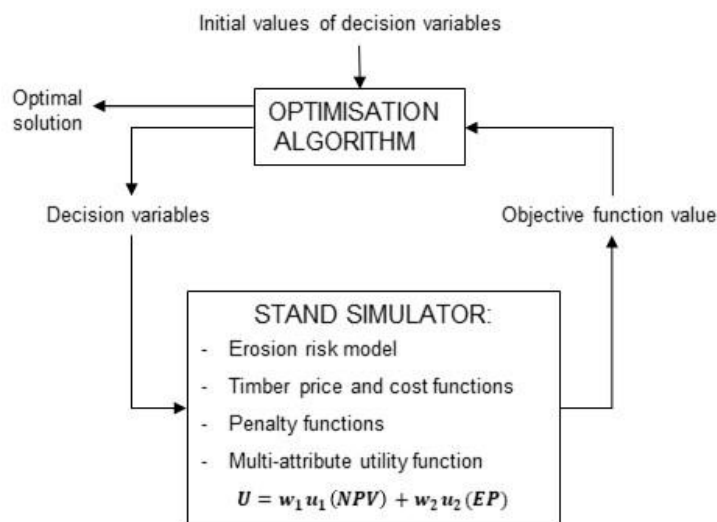
Figure 4: Structure of the simulation optimisation system.

Multi-attribute utility theory was applied to evaluate management alternatives (Pukkala 2002). The method allowed evaluations with different weights on NPV and EP, measured in different units. The original units were converted into sub-utilities, which ranged from zero to one. Production possibility boundaries for NPV and EP were defined by the criterion weights of the sub-utilities. All solutions were Pareto-optimal, meaning that it is not possible to increase a criterion variable without decreasing the other (Branke et al. 2008). Therefore, the calculated production possibility boundaries show the trade-offs between NPV and EP. Five combinations of weights for NPV and EP were used: 1/0, 0.75/0.25, 0.5/0.5, 0.25/0.75 and 0/1 for NPV and EP, respectively. The general form of additive multi-attribute utility function was as follows:

$$U = \sum_{i=1}^{n} w_i u_i(x_i) \tag{3}$$

where U is the total utility, n is the number of criteria, w_i is the weight of criterion i, u_i is the sub-utility function of criterion i and x_i is the quantity of criterion variable i.

The economic data included timber prices and harvesting and transportation costs. They were incorporated into calculations as functions where the unit price or cost depended on the diameter of the harvested tree. Net Present value (NPV) was used to evaluate the economic profit from timber production. NPV takes into account all future revenues and costs, discounted to the present time. The discount rate was 2% in all scenarios. For uneven-aged steady-state forests, NPV can be calculated from the following formula:

$$NPV_t = \frac{VG_t}{((1+i)^t - 1)} \tag{4}$$

where NPV_t is the net present value for the t-year cutting cycle (€ ha^{-1}), VG_t is the net income obtained from trees harvested every t years (€ ha^{-1}), i is the rate of interest (percentage divided by 100), and t is the length of the cutting cycle.

RESULTS

Factors related to surface erosion at the stand level

The site and stand characteristics of the eroded plots were different between the four forest types. The mean values of site and stand variables had significant differences. In particular, plots on semiarid forest lands had clear differences in some variables (Fig. 5). Semiarid forest lands are characterised in general by lower stand densities, higher shrub cover and a higher percentage of dead trees compared to forest types located in more humid areas. The mean values of the variables gave a general direction for the differences between non-eroded and eroded plot characteristics, whereas the CART method provided deeper insight into the effect of forest structure, revealing forest structures prone to erosion.
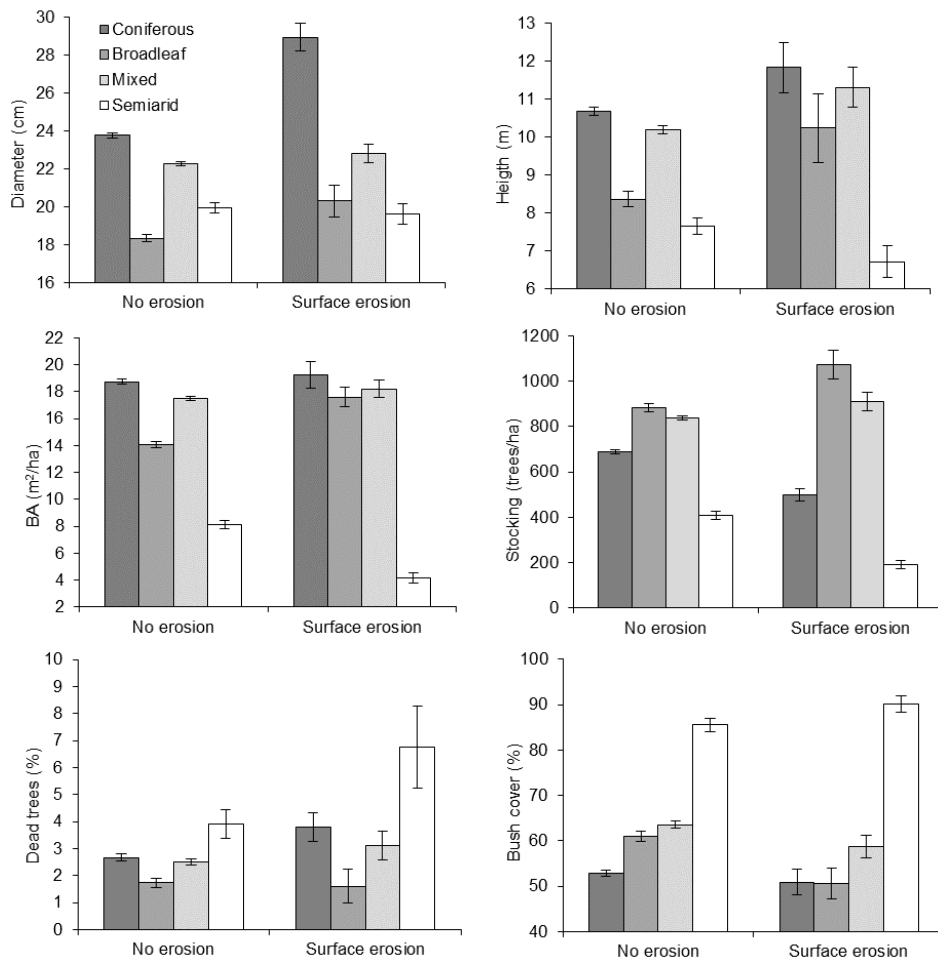
Figure 5: Mean values of forest structural variables by forest type in non-eroded plots and plots with surface erosion. The error bars represent the 95% confidence intervals.

Classification trees for four forest types show the structural characteristics that differentiate forest stands in their erosion probability (Fig. 6). For pure coniferous and broadleaf forest as well as mixed forest the best variable for bisecting the data was the tree species. Forest plots dominated by *Abies alba* and *Fagus sylvatica* had a higher presence of surface erosion compared to plots dominated by other species. In both cases, the erosion probability was higher in stands with large trees, as the splitting values of height, diameter and basal area indicate. For other species, later splits were created by tree height, stand basal area and stocking, bisecting the data into more and less erosion risky plots. In pine-dominated forest, the splits indicated that sparse forest on steeper slopes, specifically with many dead trees, was more prone to erosion. Broadleaf and mixed forest on steep slopes over 25° with high stand density had high erosion probability. Soil type was used as splitting variable in coniferous (*cambisol*) and semiarid (*fluvisol*) forest types. In addition, treatment (e.g. thinning) was found to be a splitting variable in broadleaf and mixed forest, where treated plots had a higher erosion occurrence. In semiarid areas, sparse forest on steep slopes and sparse forest on *fluvisol* soil type had the highest erosion probability.

The accuracy of the classification trees was evaluated by the area under the ROC curves (AUC). The AUC values were 0.78, 0.73, 0.66 and 0.73 for pure broadleaf, pure conifer, mixed and semiarid forest, respectively. The map of the erosion probability for the Catalonian region (Fig. 7) was drawn based on the erosion probabilities from the classification tree results. The NFI plots were classified into three erosion risk classes: low risk <0.1, moderate risk 0.1-0.4 and high risk >0.4. (Fig 7). There were no plots with probabilities between 0.29 and 0.4, therefore the high risk value was set at >0.4.
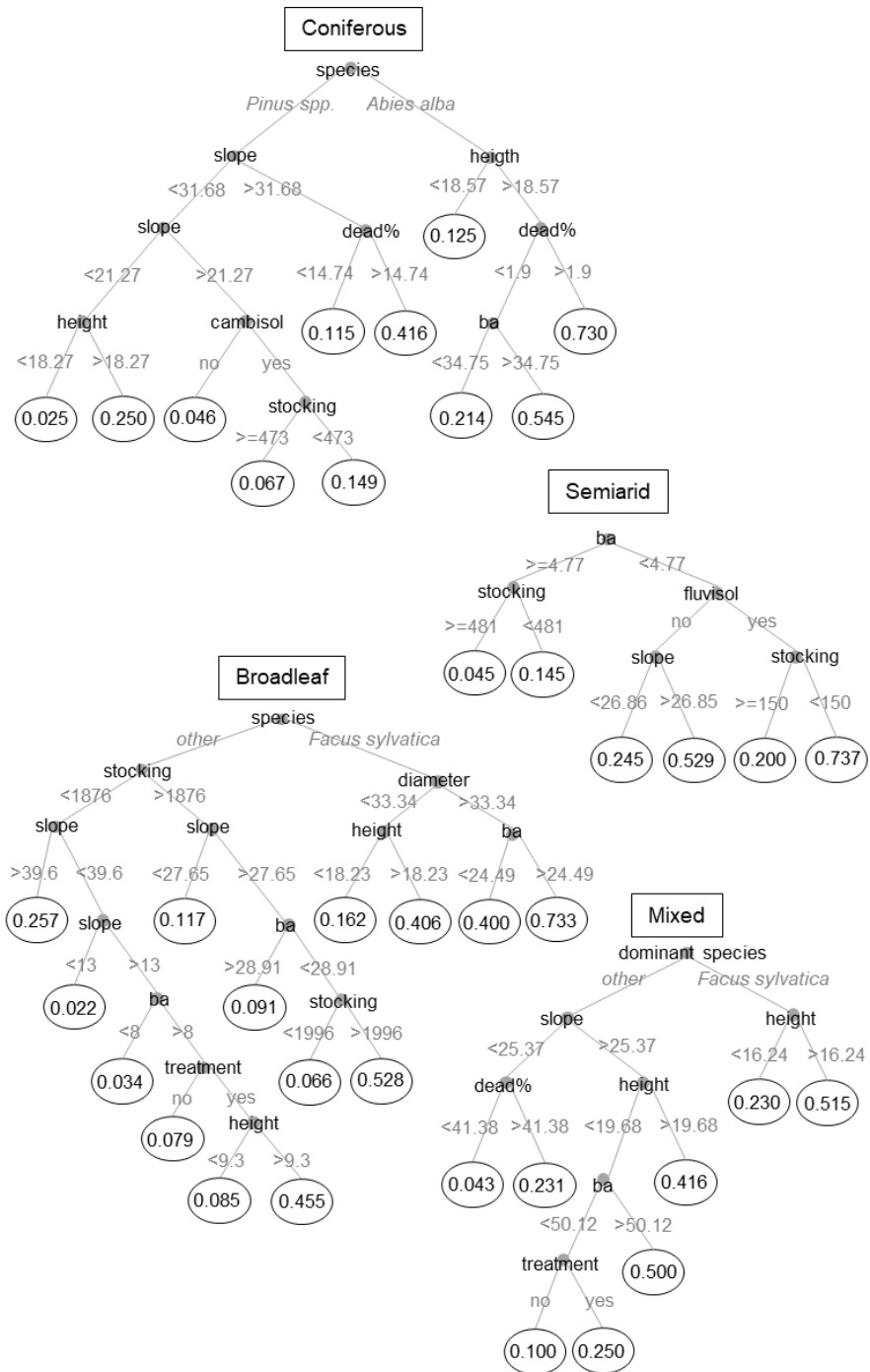
Figure 6: Classification trees for pure coniferous, pure broadleaf, mixed and semiarid forest types, with cutting values of bisecting variables. Erosion probabilities are shown at the ends of the leaves.
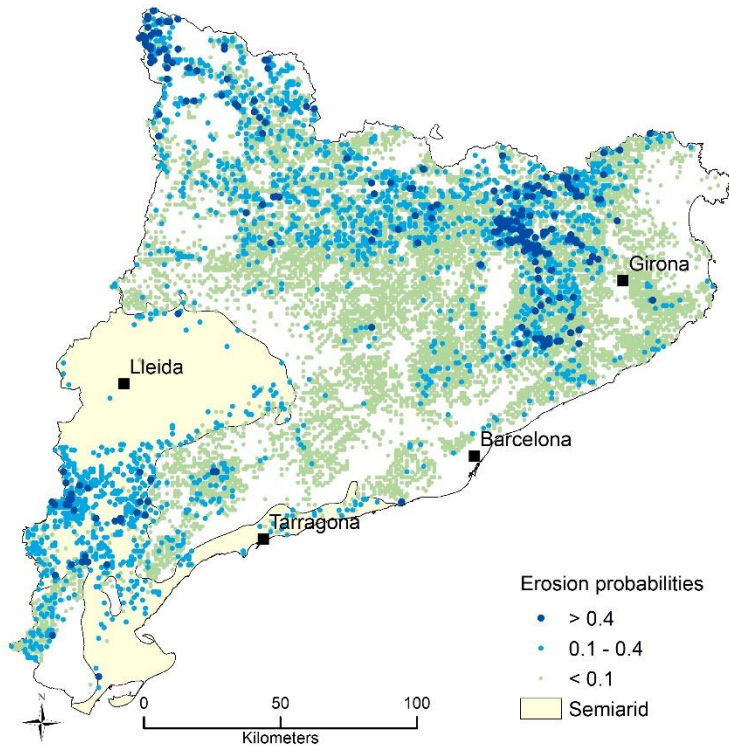
Figure 7: Probability of surface erosion risk on NFI plots in Catalonia.

Factors related to gully erosion at landscape level

The regional analysis of study **II** of factors related to the occurrence of gully erosion included both plot- and landscape-level variables. The first assessment examined differences in land-use coverage and their changes between 1987 and 2005. The land-use coverage and change analysis showed that the main land-use types in the drainage areas were forest and shrubland. The results show significant differences in land use between the drainage areas defined for gullied and non-eroded plots. The drainage areas of gullied plots had a higher percentage of shrublands, agriculture, orchards, urban and bare land, whereas non-eroded plots had a higher percentage of forest cover (Fig. 8).

The amounts and shares of road types were different in the drainage areas of gullied and non-eroded plots. Most of the non-eroded plots (~65%) had no roads in their drainage areas, whereas most of the drainage areas of gullied plots (~64%) contained unpaved roads. Typically, unpaved roads are forest and agricultural roads, often lacking proper drainage systems. Paved roads usually have engineered drainage systems for directing the surface water flow that might otherwise induce erosion.
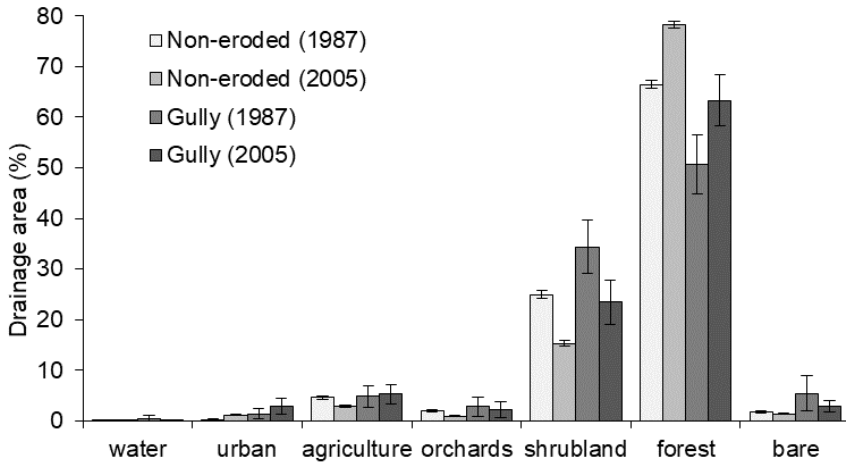
Figure 8: Average land-use coverages on non-eroded and gullied drainage areas.

The second part of study **II** assessed the probability of gullies through binary logistic regression. The smallest Akaike information criterion (AIC) value was obtained for the model including nine predictor variables:

$$P_{gully} = \left(1 + e^{-[-6.827+0.699A+0.812Dslope+2.46urban-0.049BA+2.396AND+0.757CAM+0.594fine-1.453Proad+0.636UProad]}\right)^{-1}$$

(5)

where P_{gully} is the probability of gully erosion occurrence, A is drainage area size (ha), *Dslope* is the slope of the drainage area along the stream line (degrees), *urban* is the relative increase in urban land use, *BA* is the basal area of the NFI plot (m^2 ha^{-1}), *AND* is the soil type andosol, *CAM* is the soil type cambisol, *fine* is the soil texture type, *Proad* is paved road and *UProad* is unpaved road. All variables included in the model were significant according to the Wald test ($p<0.05$). The Nagelkerke R Square was 0.304, the model classification accuracy was 0.98 and the AUC value was 0.938.

According to the model, an increase in basal area and the presence of paved roads reduced gully erosion occurrence. Variables associated with gully erosion at the plot level were the fine-textured soil types *andosol* and *cambisol*, which are prone to gully erosion on slopes. Drainage area characteristics related to gully erosion occurrence were the size of the drainage area and the steepness of the slope. When the size of the drainage area increases, water is collected from larger areas. Increasing slope increases the erosive power of running water. Furthermore, expansion in urban land use and the presence of unpaved roads increased gully erosion occurrence. Both of these are factors associated with changes in the local hydrology, affecting the infiltration capacity and surface run-off.

Trade-off between erosion risk, timber profit and structural diversity

Integrating the erosion risk into forest management and planning was studied by optimising the management of uneven-aged *Abies alba* stands using the multi-objective optimisation method (**III**). The trade-offs between three ES, namely timber production, erosion protection and biodiversity, were visualised using production possibility boundaries. The results revealed relevant trade-offs between the profit from timber production and erosion protection, depending on the slope steepness and the length of the cutting cycle. Biodiversity constraint reduced both timber profit and erosion protection values, as a proportion of the large trees remained in the forest. As the slope increased, the profit from timber production reduced due to increasing harvesting and transportation costs (Figure 1 in study **III**).

The production possibility boundaries showed the increasing influence of NPV on erosion protection (Fig. 9). Longer cutting cycles decreased NPV. The concave form of the production possibility boundaries suggests increasing rates of transformation, especially on steep slopes. The erosion protection values were constantly high on 10% slopes. Including biodiversity constraint as an obligatory objective resulted in production possibility boundaries closer to the origin, which means that scenarios including biodiversity always resulted in lower NPV and erosion protection. In general, the effect of NPV on erosion protection was marginal on a 10% slope and large on an 80% slope.
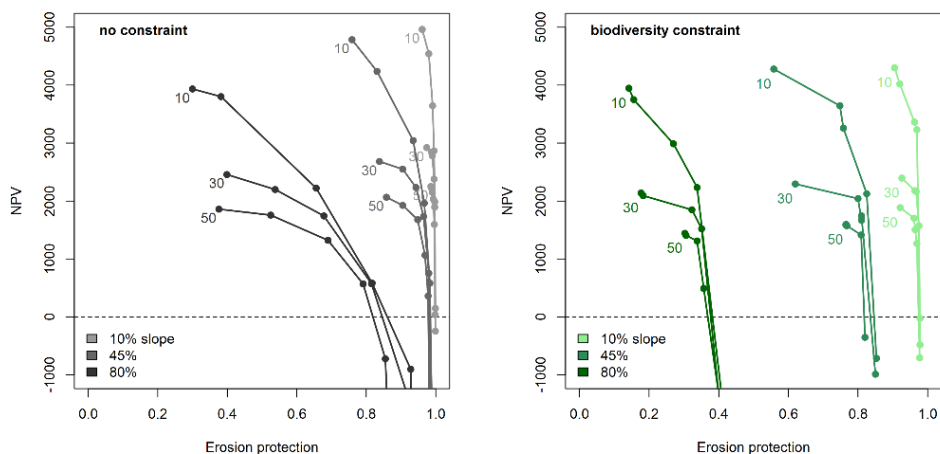
Figure 9: Production possibility boundaries for three slopes and 10-, 30- and 50-year cutting cycles representing the trade-offs between erosion protection and net present value (NPV) a) without and b) with biodiversity constraint.

The assessment of trade-offs between forest ES can help in the decision-making where different management alternatives can be implemented. The optimal weights of management objectives depend on the priorities set by the forest owner. As most *Abies alba* stands grow on steep slopes, erosion protection should be considered in many stands. The illustration of the distribution ranges of *Abies alba* stands on different slope steepness classes (according to NFI plots) in the Spanish Pyrenees demonstrates the magnitudes of weights that might be given to different management objectives (Fig. 10). The results suggest that defining proper weights for ES is important to satisfying management objectives in a sustainable way in multi-functional mountain forests, and these weights should be site-specific.
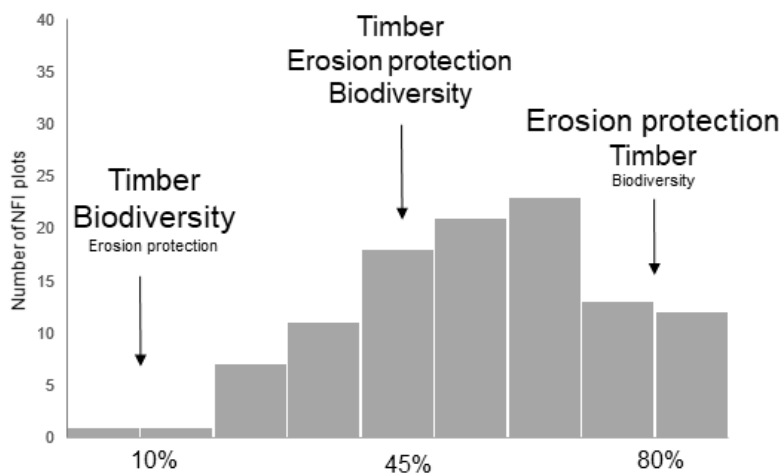
Figure 10: Illustration of the distribution of *Abies alba* on different slope steepness classes and suggested weights of three ecosystem services (timber, erosion protection and biodiversity) based on the results (font size reflects the weights of the management objectives).

DISCUSSION

The present thesis aimed at developing modelling tools for assessing soil erosion on forest lands based on forest variables, in order to integrate erosion risk assessment into forest management and planning. The thesis focused on the relation between erosion occurrence and influencing factors such as forest structure and composition, site and land-use characteristics. The results of these relationships were used to integrate erosion protection as an objective in multi-functional forest management, using an optimisation and simulation system.

The erosion observations and tree measurements were retrieved from Spanish NFI records, which provided a large amount of data for modelling, covering the forest lands of the Catalonian region. The studies in this thesis are probably the first regional-level erosion research based on actual erosion observations using an extensive and systematic dataset. Traditionally, most erosion studies on forestlands have been based on experimental plots (e.g. Edeso et al. 1999; Romero-Diaz et al. 1999, Johansen et al. 2001; Hartanto et al. 2003) or small-scale watersheds (e.g. Boix-Fayos et al. 2008; Casalí et al. 2010), therefore restricting the extent of the application area. Large-scale erosion studies have been mainly based on empirical models, for example USLE and RUSLE, to estimate the potential sediment loss of larger areas, or even at the EU level (Cerdan et al. 2010; Panagos et al. 2015; Borrelli et al. 2016).

In addition, the combined use of erosion observations and forest data retrieved from the NFI made it possible to relate forest structural variables to the presence of erosion, therefore enabling the assessment of erosion risk related to forest management alternatives. NFI data have been used in a similar way for assessing natural disturbances, for characterising stands vulnerable to damage and for creating predictive models (e.g. for snow and wind in Valinger and Fridman 1999; Martín-Alcón et al. 2010; Díaz-Yáñez et al. 2019; for fire in González et al. 2006; for moose damage in Nevalainen et al. 2016; Díaz-Yáñez et al. 2017). However, while these disturbances have direct effects on the trees, erosion has a different nature. Erosion processes typically slowly reduce the productivity of land by decreasing nutrient and water availability, which can lead to forest degradation affecting tree growth in the long term.

However, the studies presented in this thesis also have obvious limitations. The records of erosion provided by the NFI, while extensive, were based on visual assessments. This adds a subjective component to the data that can lead to possible biases. Moreover, the erosion records were only available at the time of the studies for a single NFI, which precludes confirmatory analysis or studies on the temporal development of erosion processes. Variables traditionally measured in erosion studies, such as the amount of sediment loss (e.g. Edeso et al. 1999), were not available. Although the exact amount of the sediment loss cannot be estimated, identifying the erosion class provided some indication of its scale. It can be assumed that surface erosion delivers smaller and gullies larger quantities of sediments.

Assessment of erosion risk in forest stands

The assessment of erosion occurrence was based on the available observations of the two most common erosion types: surface erosion, in order to assess erosion occurrence at the stand level (**I**), and gully erosion, in order to assess erosion at the landscape level (**II**). Erosion assessment at the regional level revealed that several forest site and stand variables were related to surface erosion on Catalonian forest lands. The mean values of the stand variables

gave a general direction for the differences in stand characteristics between non-eroded and eroded plot, whereas the CART method provided deeper insight into the forest structures where the erosion occurrence was high. The results showed two main trends related to erosion on forest lands. First, erosion seems to be due to a lack of protective vegetation mainly in dryer semiarid areas and pine forests. The second trend suggested that erosion-related risk is high in two tree species, *Abies alba* and *Facus sylvatica,* in over-stocked stands with large trees located in the Pyrenean mountains in high-precipitation areas.

Several of the existing erosion models used for forest lands are based on empirical models developed to predict sediment losses on agricultural lands in the USA (e.g. USLE, RUSLE). Therefore, they often overestimate or underestimate the sediment losses on forest lands and ignore forest structural variables (Laflen et al. 2004; Croke and Nethery 2006). Mediterranean forest environments are extremely variable in both their soil and vegetation structure. Shrub vegetation under the forest canopies in particular can have high spatial variation, which affects the potential source of sediments on forest lands (Scarascia-Mugnozza et al. 2000; Gracia et al. 2004; Palahí et al. 2008). In addition, the protective function varies between shrub and tree species due to morphological characteristics such as the leaf form and height of the plants (Martínez Raya et al. 2006). This causes differences in the rainfall partitioning, water interception, throughfall and stem flow. These effects also depend on tree age (Durán Zuazo et al. 2008; Barbier et al. 2009).

Recent studies have suggested that several forest variables could be related to erosion sensitivity, for instance stand density (Razafindrabe et al. 2010), tree height (Brandt 1987; Calder 2001) and tree species (Barbier et al. 2009). Besides this, the successional state of the forest has been found to play a role in erosion protection, so that younger forests deliver smaller amounts of sediments than older forests (Geissler et al. 2012). All these forest variables were found to be related to surface erosion in this thesis. The results revealed that erosion probability is high in areas of sparse forest cover but also in areas of dense forest where stand structural variables, tree height, basal area, mean diameter and stocking were related to erosion probability. In addition, tree species, namely the presence of *Abies alba* and *Facus sylvatica,* were also related to higher erosion probability. Both of these are shadowing tree species growing on steeps slopes in high-precipitation areas, explaining why the erosion risk is high in these forests. Dense stands with large tall trees reduce the understory vegetation cover, the amount of which correlated negatively with the stand basal area (Coll et al. 2011). Lack of understorey vegetation leaves the soil surface less protected against splash erosion induced by raindrops falling from high tree canopies, as well as the erosive effect of surface water flowing down the slopes.

Several studies have found that forest fires increase erosion probability (e.g. Andreu et al. 2001; Johansen et al. 2001; Fox et al. 2006; Shakesby 2011). In this thesis, forest fires were not, surprisingly, related to soil erosion at the stand (study **I**) or landscape level (study **II**). However, on semiarid forest land the eroded plots had the highest mean percentage of dead trees, possibly affected by fire, but the fire might have occurred earlier than 1989 which was the earliest recorded fire year on the fire perimeter maps. In addition, in semiarid areas *Pinus halepensis* regenerates quite rapidly after fire (Pausas et al. 2004b). Several studies have reported increased sediment supply from surface erosion and gully erosion after forest fires (e.g. Neary et al. 2005; Andreu et al. 2009; Cannon et al. 2010). It could be considered that the time since a fire was too long, as erosion rates are typically high immediately after a fire, after which the vegetation recovers quickly (DeBano et al. 1998; Cerda and Doerr 2005;

Ferreira et al. 2009). Fire severity, i.e. how much vegetation is left (Bastos et al. 2011), climatic conditions, and timing and intensity of rainfall (Andreu et al. 2001) also affect erosion rates after forest fires.

Landscape-level factors related to erosion

The regional-level analyses of the gullied and non-eroded drainage areas (study **II**) revealed that there are several environmental factors related to gully erosion occurrence depending on both local factors as well as the surrounding environment and land use. The results were consistent with previous studies that associated the size and slope of the drainage area (e.g. Vandekerckhove et al. 2000) and land-use changes (e.g. Vanwalleghem et al. 2003; Lesschen et al. 2007) with increasing gullying processes. Land use and vegetation types are strongly related to the variation in run-off and sediment yields in watersheds (Vandekerckhove et al. 2000). Changes in land use, abandonment of traditional practices in agriculture and forestry and increase in urbanisation often particularly increase erosion rates (Nelson and Booth 2002; Poesen et al. 2003; López-Moreno et al. 2006; Leh et al. 2013). An increase in forested area can be explained by the reforestation policy which has notably increased the forested areas in the region. The expansion of forested area since the 1950s in Catalonia has been mentioned in several other studies (e.g. Ameztegui et al. 2015; Cervera et al. 2019). In addition, land-use changes to vineyards (Martínez-Casasnovas et al. 2009) and abandonment of agricultural lands (García-Ruiz et al. 2005) have been associated with increased sediment losses from gullying processes. The results related the increase of the urban areas to gully erosion occurrence, as urban construction typically causes changes in the local hydrology, reducing soil infiltration capacity, increasing surface run-off and increasing the soil erosion risk near urban areas (Junior et al. 2010; Leh et al. 2013). Enlargement of urban areas is one of the main reasons for increased erosion risk in the future. This should be considered in urban planning. Adequate management actions should be taken to mitigate the erosion risk. In addition, the presence of unpaved roads was related to gully erosion. Road construction can change the hydrology of the area. Erosion typically starts from road curves and culverts, where water runs off the road and can induce gully erosion especially on unpaved roads, as was shown in study **II** and observed in other studies (e.g. Croke and Mockler 2001; Nyssen et al. 2002; Jordan and Martínez-Zavala 2008).

The future climate change projections for the Mediterranean predict lower rainfall, more droughts and higher potential evapotranspiration, which would decrease run-off and therefore reduce water erosion (Nunes et al. 2008; IPCC 2013). On the other hand, extreme climatic conditions might become more frequent, with heavier rainfall intensities, which in turn could actually produce greater sediment losses when the vegetation cover has suffered from drought (Nearing et al. 2005; IPCC 2013). Besides climate change, even greater soil erosion potential is related to changes in land use (Nearing et al. 2005; Serpa et al. 2015).

Integrating ecosystem services into forest management

The erosion models were the basis for integrating erosion risk into forest management planning (**III**). For this study, *Abies alba* was selected, as previous studies showed that this species was associated with high erosion probabilities and there are clear relationships between

surface erosion and stands' structural variables. Therefore, these forests serve as a good example for the last study.

Forests provide several ecosystem services whose importance has recently increased, leading to increasing complexity of forest management and planning with multiple objectives (Duncker et al. 2012). Forest management should aim to enhance the provisioning of several ES, including soil protection and water regulation. At the same time, the effects of climate change and its influence on ES has been acknowledged (Morán-Ordóñez et al. 2019). Several studies have incorporated the provisioning of different goods and services to the optimisation of forest management. For example, the joint production of timber and mushrooms (Palahí et al. 2009), timber and berries (Peura et al. 2016; Kurttila et al. 2018), timber and pine honey (de Miguel et al. 2014) and timber and pine seeds (Pasalodos-Tato et al. 2016) have been optimised. In addition, the optimal forest management for ES which do not directly produce economic income has been studied, for example biodiversity (Nelson et al. 2009; Mönkkönen et al. 2014) and protection against landslides and rockfalls (LaFond et al. 2017). Studies on soil protection as one ES are rare. However, the topic has gained importance recently, and some surveys of forest ES including soil protection have been undertaken (e.g. Roces-Díaz et al. 2018) where the erosion protection has been defined as constant if forest cover exists on slopes steeper than 30%. In this thesis, the erosion risk was related to the stand structural variables and influence of the slope, which made it possible to integrate erosion protection as one forest ES into forest management using simulations and optimisation.

Managing forests for multiple purposes can be complex, since maximising the production of one service often results in a decline of another service, meaning that there are trade-offs between services (Bennet et al. 2009). The use of simulation and optimisation can help to define proper weights for the management objectives and evaluate trade-offs and the feasibility of obtaining several ES simultaneously. In this thesis (study **III)**, this was demonstrated by studying the trade-offs between the economic profit from timber in *Abies alba* stands while controlling the erosion risk and maintaining biodiversity. In the case of *Abies alba* in the Pyrenean mountains, studies show that many stands are over-stocked (Aunos and Blanco 2006; Oliva and Colinas 2007). Land abandonment and rural migration to urban areas are partial reasons for the existence of these dense stands where the lack of forest management is threating regeneration and increasing tree mortality (Oliva and Colinas 2007). Increasing stand density and lack of regeneration may lead to increased erosion. Forest management, especially in mountain forest, should be regionally adapted to site and stand conditions due to the high variability in environmental conditions (Mina et al. 2017). Forest owners should carefully choose their management strategies: whether the production of several ecosystem services simultaneously is feasible, or whether it would be better to modify management objectives according to site and stand conditions.

Erosion risk was found to be highly dependent on site characteristics and stand structure, which can be modified through management. In the case of *Abies alba* stands as an example, the assumption that denser forest provides more erosion protection was found to be not fully true. Therefore, the results would be different, for example, in *Pinus spp.* forest, where study **I** indicated that pine forest is prone to erosion when the stand density is low on *cambisol* soils on slopes and when the percentage of dead trees is high, especially on steep slopes. As the results show, the erosion protection should be evaluated separately for different forest types, as the relation between soil protection and structural characteristics can be very different.

Managing forest for several forest ES can result in different and often contradicting management practices. Using the plant cover for soil protection can represent efficient management when reducing the erosion is the sole objective, but on the other hand it might affect

the production of the other ES. For example, vegetation cover between vine rows has reduced sediment losses in a semiarid area but also decreased the productivity of vineyards due to competition for water (Marques et al. 2010). In addition, increasing vegetation cover increases the risk of forest fires in fire-prone areas. Forest management practices for reducing the fire risk include reducing the multi-layered structure of the stand and creating fire breaks where the fuel material is removed, leaving the soil vulnerable to erosion processes (Vélez 1986). In the simulated cases (**III**) when the biodiversity objective was included, leaving a minimum of ten large trees in the stand had a negative impact on erosion protection on steep (80%) slopes. Therefore, the priorities for obtaining different ES must be decided separately for each site.

Analysing the forest site and stand characteristics related to soil erosion can help to recognise forests susceptible to erosion and determine management practices that can enhance soil protection while considering other forest ES simultaneously. Study **III** aimed at incorporating erosion risk into forest management at stand level. In the future, landscape-level studies should be conducted to analyse erosion protection as one forest ES at a larger scale, and for other forest types.

Future prospects

The main aim of this thesis was to assess soil erosion on forest lands at the regional level, by characterising variables related to erosion occurrence and integrating the estimated erosion risk into forest management planning. Erosion processes have often been studied on small scales, based on expensive and time-consuming field experiments or on sediment loss estimations using existing erosion models. The availability of the extensive records of visual evaluations of erosion states in the National Forest Inventory made it possible to connect the erosion location and state of erosion to forest variables, allowing for assessment at the regional level. This approach can be a starting point towards more ambitious objectives, enlarging the area studied and the time period analysed.

Concerning the area studied, the approach and methods presented can be applied to other Spanish regions with similar data available, as this could help identify additional interactions and trade-offs not contemplated in this study. As only two erosion types were considered, future studies should certainly expand the approach taken by including rill erosion and landslides. This must imply additional compilation efforts to create a comprehensive database with different erosion measurements, for example, experimental plot- and basin-level sediment measurements. Concerning expanding the timeframe of the study, the release of the fourth Spanish National Forest Inventory can offer the possibility to study the impact of forest evolution on erosion and can help to validate the models proposed.

In addition, the development of remote sensing techniques has opened up the opportunity to conduct large-scale assessments using digital terrain models and satellite images with high resolution, that can be combined with the visual records available. In particular, accurate digital elevation models (DEM) with high spatial resolution generated from airborne laser scanning (ALS) data at different time intervals offer the possibility to detect large gullies and landslides under forest canopies (e.g. James et al. 2007) which, combined with NFI-based observations, could offer new ways to conduct erosion studies on forest lands.

From the modelling perspective, the erosion protection as a forest ES and its trade-offs with other ES could also be studied at a more ambitious forest and landscape level, adding complexity to the models and approach taken. For instance, increasing the number of risks

simultaneously, by combining, for example, fire and erosion, in addition to other forest eco-system services such as carbon sequestration or non-wood products, based on Pareto Frontier methods (e.g. Borges et al. 2014; Garcia-Gonzalo et al. 2015) could result in a more inte-grated view of the risk. This could be implemented at both the forest and landscape level, greatly expanding the ambition of the current study, helping to identify where and which specific management practices must be allocated and could help better mitigate the erosion risk on forest lands.

REFERENCES

Ameztegui A., Coll L., Brotons L., Ninot J.M. (2015). Land-use legacies rather than climate cange are driving the recent upwards shift of the mountain tree line in the Pyrenees. Global Ecology and Biogeography 25: 263-273.
https://doi.org/10.1111/geb.12407

Anaya-Romero A., Muñoz-Rojas M., Ibáñez B., Marañon T. (2016). Evaluation of forest ecosystem services in Mediterranean areas. A regional case study in South Spain. Ecosystem services 20: 82-90.
http://dx.doi.org/10.1016/j.ecoser.2016.07.002

Andreu V., Imeson A.C., Rubio JL. (2001). Temporal changes in soil aggregates and water erosion after a wildfire in a Mediterranean pine forest. Catena 44: 69-84
https://doi.org/10.1016/S0341-8162(00)00177-6

Aunos A. & Blanco R. (2006). Caracterización de rodales irregulars de *Abies alba* Mill. Capitalizados en exixtencia del Valle de Arán (Pirineos, España) y de las cortas de entresaca practicadas. In

Barbier S., Balandier P., Gosselin F. (2009). Influence of several tree traits on rainfall partitioning in temperate and boreal forests: a review. Annals of Forest Science 66: 602.
DOI: 10.1051/forest/2009041

Bare B., Opalach D. (1988). Determining investment-efficient diameter distributions for uneven-aged northern hardwoods. For. Sci. 34: 243–249.

Bastos A., Gouveis C.M., DaCamara C.C., Trigo R.M. (2011). Modelling post-fire vegetation recovery in Portugal. Biogeosciences 8: 3593-3607.
doi:10.5194/bg-8-3593-2011

Bennett E.M., Peterson G.D., Gordon L.J. (2009). Understanding relationships among multiple ecosystem services. *Ecol. Lett.* 1:, 1394–1404.
https://doi.org/10.1111/j.1461-0248.2009.01387.x

Blanco E., Casado M.A., Costa M., Escribano R.,García M., Génova M., Gómez F., Moreno J.C., Morla C., Regato P., Sáinz H. (1997). Los bosques ibéricos: Una interpretación geobotánica. Planeta, Barcelona.

Blanco H. & Lal R. (2008). Principles of soil conservation and management. Springer, Netherlands.

Bochet E., Poesen J., Rubio JL. (2006). Runoff and soil loss under individual plants of a semi-arid Mediterranean shrubland: influence of plant morphology and rainfall intensity. Earth Surface Processes and Landforms 31:536-549.
https://doi.org/10.1002/esp.1351

Boer M., Puigdefábregas J. (2005) Effects of spatially structured vegetation patterns on hillslope erosion in a semiarid Mediterranean environment: a simulation study. Earth Surface Processes and Landforms 30: 149-167.
https://doi.org/10.1002/esp.1180

Boix-Fayos C., de Vente J., Martínez-Mena M., Barberá G.G., Castillo V. (2008). The impact of land use change and check-dams on catchment sediment yield. Hydrological Processes 22: 4922-4935.
DOI: 10.1002/hyp.7115

Borrelli P., Panagos P., Langhammer J., Apostol B., Schütt B. (2016). Assessment of the cover changes and soil loss potential in European forestland: First approach to derive

indicators to capture the ecological impacts on soil-related forest ecosystems. Ecological Indicators 60: 1208-1220.
https://doi.org/10.1016/j.ecolind.2015.08.053

Brandt J. (1987). The effect of different types of forest management on the transformation of rainfall energy by the canopy in relation to soil erosion. In: Proceedings of the forest hydrology watershed management conference Vancouver, BC (1987), IAHS Publication No. 167: 213–222.

Branke J., Deb K., Miettinen K., Slowiński R. (2008). Multiobjective optimization- Interactive and evolutionary approaches. Springer, Heidelberg.

Breiman L., Friedman J.H., Olshen R.A., Stone C.J. (1984). Classification and regression trees. Wadsworth International Group, Belmont

Calder I.R. (2001). Canopy processes: implications for transpiration, interception and splash induced erosion, ultimately for forest management and water resources. Plant Ecology 153:203–214.
https://doi.org/10.1023/A:1017580311070

Cannon S.H., Gartner J.E., Rupert M.G., Michael J.A., Rea A.H., Parrett C. (2010). Predicting the probability and volume of postwildfire debris flows in the intermountain western United States. GSA Bulletin 122: 127–144.
DOI:10.1130/B26459.1.

Casalí J., Giménez R., Díez J., Álvarez-Mozos J., Del Valle de Lersundi J., Goñi M., Campo M.A., Chahor Y., Gastesi R., López J. (2010). Sediment production and water quality of watersheds with contrasting land use in Navarre (Spain). Agricultural Water Management 97: 1683-1694.
doi:10.1016/j.agwat.2010.05.024

Cerda A. & Doerr S.H. (2005). The influence of vegetation recovery on soil hydrology and erodibility following fire: an eleven year investigation. Int. J. of Wildland Fire 14: 423–437.

Cerdan O., Govers G., Le Bissonnais Y., Van Oost K., Poesen J., Saby N., Gobin A., Vacca A., Quinton J., Auerswald K., Klik A., Kwaad F.J.P.M., Raclot D., Ionita I., Rejman J., Rousseau S., Muxart T., Roxo M.J., Dostal T. (2010). Rates and spatial variations of soil erosion in Europe: a study based on erosion plot data. Geomorhology 122:167-177

Cervera T., Pino J., Marull J., Padró R., Tello E. (2019). Understanding the long-term dynamics of forest transitions: From deforestation to afforestation in a Mediterranean landscape (Catalonia, 1868-2005). Land Use policy 80: 318-331.
https://doi.org/10.1016/j.landusepol.2016.10.006

Clavero P., Martín Vide J., Raso J.M. (1997). Atles climátic de Catalunya (ACC). Institut Cartogràfic de Catalunya i Departament de Medi Ambient, Generalitat de Catalunya, Barcelona

Coll L., González-Olabarria J.R., Mola-Yudego B., Pukkala T., Messier C. (2011). Predicting understory maximum shrubs cover using altitude and overstory basal area in different Mediterranean forest. Eur J For Res 130: 55–65. DOI 10.1007/s10342-010-0395-y

Costa Freitas M.B., Xavier A., Fragoso R. (2017). Integration of fire risk in a sustainable forest management model. Forest 8: 270. doi:10.3390/f8080270

Croke J. & Mockler S. (2001). Gully initiation and road-to-stream linkage in a forested catchment, southeastern Australia. Earth Surface Processes and Landforms 26: 205-217. doi:10.1002/1096-9837(200102)

Croke J., Hairsine P., Fogarty P. (2001). Soil recovery from track construction and harvesting changes in surface infiltration, erosion and delivery rates with time. Forest Ecology and Management 143, 3-12. https://doi.org/10.1016/S0378-1127(00)00500-4

Croke J. & Nethery M. (2006). Modelling runoff and soil erosion in logged forest: Scope and application of some existing models. Catena 67:35-49. https://doi.org/10.1016/j.catena.2006.01.006

DeBano L. F., Neary D. G., Folliott P. F. (1998). Fire's Effects on Ecosystems, John Wiley and Sons, New York, 333 pp.

De la Base Topográfica. (2007). 1:50 000 version 3·1. BT-50M, Institut Cartografic de Catalunya. Vector map.

de Figueiredo T., Fonseca F., Martins A. (2012). Soil loss and run-off in young forest stands as affected by site preparation technique: a study in NE Portugal. European Journal of Forest Research 131:1747-1760. https://doi.org/10.1007/s10342-011-0581-6

de Miguel S., Pukkala T., Yeşil A. (2014). Integrating pine honeydew production into forest management optimization. European Journal of Forest Research. 133, 423-432. https://doi.org/10.1007/s10342-013-0774-2

DGCN (2002). Plan Forestal Español 2002–2032. Ministerio de Medio Ambiente. Direccio´n General de Conservacio´n de la Naturaleza, Madrid.

DGCN (2005). Tercer Inventario Forestal Nacional (1997–2006) Cataluña. Ministerio de Medio Ambiente: Madrid.

Diaz-Balteiro L., Alonso R., Martínez-Jaúregui M., Pardos M. (2017). Selecting the best management alternative by aggregating ecosystem services indicators over time: a case study in central Spain. Ecological Indicators 72: 322-329. http://dx.doi.org/10.1016/j.ecolind.2016.06.025

Díaz-Yáñez O., Mola-Yudego B., González-Olabarria J.R. (2017). What variables make a forest stand vulnerable to browsing damage occurrence? Silva Fennica 51, 1-11. https://doi.org/10.14214/sf.1693

Díaz-Yáñez O., Arias-Rodil M., Mola-Yudego B., González-Olabarria J.R., Pukkala T. (2019). Simulating the effects of wind and snow damage on the optimal management of Norwegian spruce forests. Forestry 92: 406-416. https://doi.org/10.1093/forestry/cpz031

Dissmeyer G. & Foster G.R. (1980). A guide for predicting sheet and rill erosion on forest land. Technical publication. USDA Forest Service, Atlanta.

Duncker P.S., Raulund-Rasmussen K., Gundersen P., Katzensteiner K., De Jong J., Ravn H.P., Smith M., Eckmüllner O., Spiecker H. (2012). How forest management affects ecosystem services, including timber production and economic return: synergies and trade-offs. Ecology and Society 17(4): 50. http://dx. doi.org/10.5751/ES-05066-170450

Durán Zuazo V.D. & Rodríguez Pleguezuelo C.R. (2008). Soil-erosion and runoff prevention by plant covers. A reive. Agronomy for Sustainable Development 28: 65-86. http://dx.doi.org/10.1051/agro:2007062

Edeso J.M., Merino A., González M.J., Marauri P. (1999). Soil erosion under different harvesting managements in steep forestlands from Northern Spain. Land degradation and development 10 : 79-88. https://doi.org/10.1002/(SICI)1099-145X(199901/02)10:1<79::AID-LDR324>3.0.CO;2-4

Elliot W.J., Hall D.E., Graves S.R. (1999). Predicting sedimentation from forest roads. J For 97(8):23–29

EME (2011). Millennium Ecosystem Assessment of Spain—Managing Biodiversity and Ecosystem services for human well-being. *Sección III.5. Bosque y matorral esclerófilo*. Fundación Biodiversidad, Ministerio de Medio Ambiente y Medio Rural y Marino, Madrid, Spain

FAO (2018). State of Mediterranean Forest. Food and Agriculture Organization of the United Nations, Rome and Plan Bleu, Marseille. ISBN FAO: 978-92-5-131047-2

Ferri C., Flach P., Hernández-Orallo J. (2003). Decision trees for ranking: effect of new smoothing methods, new splitting criteria and simple pruning methods. Technical report, DSIC, UPV

Flanagan D.C. & Livingston S.J. (1995). Water erosion prediction project (WEPP) version 95.7 user summary. West Lafayette, IN. NSERL Report No. 11. USDA-ARS National Soil Erosion Research Laboratory, West Lafayette, Ind

Fox D., Berolo W., Carrega P., Darboux F. (2006). Mapping erosion risk and selecting sites for simple erosion control measures after a forest fire in Mediterranean France. Earth Surface Processes and Landforms 31:606-621.
DOI:10.1002/esp.1346

García-Nieto A., García-Llorente M., Iniesta-Arandia I., Martín-López B. (2013). Mapping forest ecosystem servies: from providing units to beneficiaries. Ecosystem Services 4:126-138. http://dx.doi.org/10.1016/j.ecoser.2013.03.003

García-Ruiz J.M., Lasanta T., Valero B., Martí C., Beguería S., López-Moreno J.I., Regüés D., Lana-Renault N. (2005). Soil erosion and runoff generation related to land use changes in the Pyrenees. Global Change and Mountain Regions 23: 321–330.
https://doi.org/10.1007/1-4020-3508-X_32

García-Ruiz J.M., Nadal-Romero E., Lana-Renault N., Beguería S. (2013). Erosion in Mediterranean Landscapes: Changes and future challenges. Geomorphology 198: 20-36.
https://doi.org/10.1016/j.geomorph.2013.05.023

Gardiner B.A., Quine C. (2000). Management of forest to reduce the risk of abiotic damage-areview with particular reference to the effects of strong winds. Forest Ecology and Management 135: 261-277. https://doi.org/10.1016/S0378-1127(00)00285-1

Geissler C., Kühn P., Böhnke M., Bruelheide H., Bruelheide H., Shi X., Scholten T. (2012). Splash erosion potential under tree canopies in subtropical SE China. Catena 91: 85-93.
doi:10.1016/j.catena.2010.10.009

Generalitat de Catalunya (1987). Classificació dels usos del sòl a Catalunya 1987. Raster map 30x30m

Gómez Gutiérrez A., Schnabel S., Lavado C.F. (2009). Gully erosion, land use and topographical thresholds during the last 60 years in a small rangeland catchment in SW Spain. Land Degradation & Development 20: 535–550.
DOI:10.1002/ldr.931.

González J.R., Pukkala T., Palahí M. (2005). Optimising the management of Pinus sylvestris L. stand under the risk of fire in Catalonia (north-east Spain). Annals of Forest Science. 62, 493-501.
DOI: 10.1051/forest:2005054

González J.R., Palahí M., Trasobares A., Pukkala T. (2006). A fire probability model for forest stands in Catalonia (north-east Spain). Annals of Forest Science, Springer Verlag/EDP Sciences, 2006, 63 (2), pp.169-176

González J.R. & Pukkala T. (2007). Characterization of forest fires in Catalonia (north-east Spain). European Journal of Forest Research 126: 421–429.
DOI:10.1007/s10342-006-0164-0.

González-Olabarria J.R., Palahí M., Pukkala T., Trasobares A. (2008). Optimising the management of Pinus nigra Arn. Stand under endogenous risk of fire in Catalonia. Inv Agrar-Sist Recursos Fores 17:10–17

Gracia C., Ibàñez J.J., Burriel J.A., Mata T., Vayreda J. (2004). *Inventari Ecològic i Forestal de Catalunya. Regió Forestal I*. CREAF, Bellaterra. ISBN: 84-931323-9-X

Haight R.G. & Monserud R.A. (1990). Optimizing any-aged management of mixed-species stands: I. Performance of a coordinate search process. *Can. J. For. Res.* **20**, 15–25.

Hartanto H., Prabhu R., Widayat A.S.E., Asdak C. (2003). Factors affecting runoff and soil erosion: plot-level soil loss monitoring for assessing sustainability of forest management. Forest Ecology and Management 180: 361–374.
DOI:10.1016/S0378-1127(02)00656-4.

Heinonen T., Pukkala T., Ikonen V-P, Peltola H., Venäläinen A., Dupont S. (2009). Integrating the risk of wind damage into forest planing. Forest Ecology and Management, 258, 1567-1577. https://doi.org/10.1016/j.foreco.2009.07.006

Hooke R. & Jeeves T.A. (1961). "Direct search" solution of numerical and statistical problems. Journal of Association for Computing Machinery. 8, 212–229. doi:10.1145/321062.321069

Hyde K., Woods S.W., Donahue J. (2007). Predicting gully rejuvenation after wildfire using remotely sensed burn severity data. Geomorphology 86: 496–511. DOI:10.1016/j.geomorph.2006.10.012.

ICONA (1991). Plan Nacional de lucha contra la erosión. Ministerio de Agricultura, pesca y alimentación. Instituto Nacional para la conservación de la naturaleza, Madrid.

IPCC (2013). Intergovernmental Panel on Climate Change, Climate Change 2013: The Physical Science Basis. In: Stocker, T.F., Qin, D., Plattner, G.-K., Tignor, M., Allen, S.K., Boschung, J., Nauels, A., Xia, Y., Bex, V., Midgley, P.M. (Eds.) Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, New York.

James L.A., Watson D.G., Hansen W.F. (2007). Using LiDAR data to map gullies and headwater streams under forest canopy: South Carolina, USA. Catena 71: 132-144. doi:10.1016/j.catena.2006.10.010

Johansen M.P., Hakonson T.E., Breshears D.D. (2001). Post-fire runoff and erosion from rainfall simulation: contrasting forest with shrublands and grasslands. Hydrological Processes. 15, 2953-2965.
DOI: 10.1002/hyp.384

Jordán A. & Martínez-Zavala L. (2008). Soil loss and runoff rates on unpaved forest roads in southern Spain after simulated rainfall. Forest Ecology and Management 255: 913–919. DOI:10.1016/j.foreco.2007.10.002

Junior O.C., Guimaraes R., Freitas L., Gomes-Loebmann D., Gomes R.A., Martins E., Montgomery D.R. (2010). Urbanization impacts upon catchment hydrology and gully development using multi-temporal digital elevation data analysis. Earth Surface Processes and Landforms 35: 611–617.
DOI:10.1002/esp.1917.

Kang W., Deng X., Zhao Z. (2008). Effects of canopy interception on energy conversion processes in a Chinese fir plantation ecosystem. Front For China 3:264–270.
https://doi.org/10.1007/s11461-008-0053-4

Kolka R.K. & Smidt M.F. (2004). Effects of forest road amelioration techniques on soil bulk density, surface runoff, sediment transport, soil moisture and seedling growth. Forest Ecology and Management 202:313-323. doi:10.1016/j.foreco.2004.07.042

Kurttila M., Pukkala T., Miina J. (2018). Synergies and trade-offs in the production of NWFPs predicted in boreal forests. Forests. 9, 417. doi:10.3390/f9070417

Laflen J.M., Elliot W.J., Flanagan D.C., Meyer C.R., Nearing M.A. (1997). WEPP-predicting water erosion using a process based model. J Soil Water Conserv 52:96–102

Laflen J.M., Flanagan D.C., Bernard A.E. (2004). Soil erosion and sediment yield prediction accuracy using WEPP. Journal of the American Water Resources Association 40: 289-297.

Lafond V., Cordonnier T., Mao Z., Courbaud B. (2017). Trade-offs and synergies between ecosystem services in uneven-aged mountain forest: evidences using Pareto fronts. European Journal of Forest Research. 136, 997-1012.
doi:10.1007/s10342-016-1022-3

Leh M., Bajwa S., Chaubey I. (2013). Impact of land use change on erosion risk: an integrated remote sensing, geographic information system and modeling methodology. Land Degradation & Development 24: 409–421.
DOI:10.1002/ldr.1137.

López-Moreno J.I., Beguería S., García-Ruiz J.M. (2006). Trend in high flows in the central Spanish Pyrenees: response to climatic factors or to landuse change? Hydrological Sciences Journal 51: 1039–1050.
DOI:10.1623/hysj.51.6.1039.

Lesschen J.P., Kok K., Verburg P.H., Cammeraat L.H. (2007). Identification of vulnerable areas for gully erosion under different scenarios of land abandonment in Southeast Spain. Catena 71: 110–121.
DOI:10.1016/j.catena.2006.05.014.

Madej M.A., Eschenbach E.A., Diaz C., Teasley R., Baker K. (2006). Optimization strategies for sediment reduction practices on roads in steep, forested terrain. Earth Surface Processes and Landforms, 31, 1643-1656. DOI: 10.1002/esp.1436

Marques M.J., García-Muñoz S., Muñoz-Organero G., Bienes R. (2010). Soil conservation beneath grass cover in hillside vineyards under Mediterranean climatic conditions (Madrid, Spain). Land Degradation & Development 21: 122-131.
DOI: 10.1002/ldr.915

Martín-Alcón S., González-Olabarria J.R, Coll L. (2010). Wind and snow damage in the Pyrenees pine forests: effect of stand attributes and location. Silva Fennica 44:399-410.
https://doi.org/10.14214/sf.138

Martínez-Casasnovas J.A., Ramos M,C., García-Hernández D. (2009). Effects of land-use changes in vegetation cover and sidewall erosion in a gully head of the Penedés region northeast Spain. Earth Surface Processes and Landforms 34: 1927–1937.
DOI:10.1002/esp.1870.

Martínez Raya A., Durán Zuazo V.H., Francia Martínez J.R. (2006). Soil erosion and runoff response to plant-cover strips on semiarid slopes (SE Spain). Land Degradation and Development 17:1-11.
DOI: 10.1002/ldr.674

MCSC (2005). Mapa de Cobertes del Sól de Catalunya. 3a edició 2005- 2007. Centre de Recerca Ecológica I Aplicacions Forestals CREAF.

MEA (2005). Millennium Ecosystem Assessment. Ecosystems and Human Wellbeing: Synthesis. Island Press, Washingthon, DC.

Merlo M. & Croitoru, L. (2005). Valuing Mediterranean forest: towards total economic value. CABI Publishing, Wallingford

Messier C., Puettmann K., Chazdon R., Andersson K.P., Angers V.A., Brotons L., Filots E., Titler R., Parrot L., Levin S.A. (2015). From management to stewardship: viewing forest

as complex adaptive systems in an uncertain world. Conserv Lett. 8: 368-377. https://doi.org/10.1111/conl.12156

Mina M., Bugmann H., Cordonnier T., Irauschek F., Klopcic M., Pardos M., Cailleret M. (2017). Future services from European mountain forest under climate change. Journal of Applied Ecology. 54: 389-401. Doi: 10.1111/1365-2664.12772

Morán-Ordóñez A., Roces-Díaz J.V., Otsu K., Ameztegui A., Coll L., Lefevre F., Retana J., Brotons L. (2019). The use of scenarios and models to evaluate the future of nature values and ecosystem services in Mediterranean forests. Regional Environmental Change 19: 415-428.
https://doi.org/10.1007/s10113-018-1408-5

Morgan R.P.C .(2005). Soil erosion and conservation, 3rd edn. Blackwell, Oxford.

Mönkkönen M., Juutinen A., Mazziotta A., Miettinen K., Podkopaev D., Reunanen P., Salminen H., Tikkanen O-P. (2014). Spatially dynamic forest management to sustain biodiversity and economic returns. Journal of environmental management. 134: 80-89.
http://dx.doi.org/10.1016/j.jenvman.2013.12.021

Nanko K., Hotta N., Suzuki M. (2004). Assessing raindrop impact energy at the forest floor in a mature Japanese cypress plantation using continuous raindrop-sizing instruments. Journal of Forest Research 9:157-164.
https://doi.org/10.1007/s10310-003-0067-6

Nearing M.A., Jetten V., Baffaut C., Cerdan O., Couturier A., Hernandez M., Le Bissonnais Y., Nichols M.H., Nunes J.P., Renschler C.S., Souchère V., van Oost K. (2005). Modeling response of soil erosion and runoff to changes in precipitation and cover. Catena 61:131-154.
doi:10.1016/j.catena.2005.03.007

Neary D.G., Ryan K.C., DeBano L.F. (eds.) (2005). Wildland fire in ecosystems: Effects of fire on soils and water. USDA Forest Service General Technical report RMRS-GTR-42-vol 4. 250 p.

Nelson E.J., Booth D.B. (2002). Sediment sources in an urbanizing, mixed land use watershed. Journal of Hydrology 264:51-68.
https://doi.org/10.1016/S0022-1694(02)00059-8

Nelson E., Mondoza G., Regetz J., Polasky S., Tallis J., Cameron D.R., Chan K.M.A., Daily G.C., Goldstein J., Kareiva P.M., Londsdorf E., Naidoo R., Ricketts T.H., Shaw M.R. (2009). Modelling multiple ecosystems services, biodiversity conservation, commodity production, and tradeoffs at landscape scale. Frontiers Ecol. Environ. 7: 4-11.
https://doi.org/10.1890/080023

Nevalainen S., Matala J., Korhonen K.T., Ihalainen A., Nikula A. (2016). Moose damage in National Forest Inventories (1986-2008) in Finland. Silva Fennica 50(2):1410.
https://doi.org/10.14214/sf.1410

Ninyerola M., Pons X., Roure J.M. (2000). A methodological approach of climatological modelling of air temperature and precipitation through GIS techniques. Int J Climatol 20:1823–184.

Nunes J.P., Seixas J., Pancheco N.R. (2008). Vulnerability of water resources, vegetation productivity and soil erosion to climate change in Mediterranean watersheds. Hydrological Processes 22: 3115-3134.
DOI: 10.1002/hyp.6897

Nyssen J., Poesen J., Moeyersons J., Luyten E., Veyret-Picot M., Deckers J., Haile M., Govers G. (2002). Impact of road building on gully erosion risk: a case study from the

Northern Ethiopian highlands. Earth Surface Processes and Landforms 27: 1267–1283. DOI:10.1002/esp.404.

Oliva J. & Colinas C. (2007). Declines of silver fir (*Abies alba* Mill.) stands in the Spanish Pyrenees: Role of management, historic dynamics and pathogens. For Ecol Manage. 252: 84-97. doi:10.1016/j.foreco.2007.06.017

Palahí M., Pukkala T. (2003). Optimising the management of Scots pine (P. sylvestris L.) stands in Spain based on individual-tree models. Ann. Sci. For. 60, 105-114. DOI: 10.1051/forest:2003002

Palahí M., Mavsar R., Gracia C., Birot Y. (2008). Mediterranean forest under focus. Int For Rev 10: 675-688.

Palahí M., Pukkala T., Bonet J.A., Colinas C., Fischer C., Martínez de Aragón J.R. (2009). Effect of the Inclusion of Mushroom Values on the Optimal Management of Even-Aged Pine Stands of Catalonia. *Forest Science*, 55, 503–511. https://doi.org/10.1093/forestscience/55.6.503

Panagos P., Borrelli P., Poesen J., Ballabio C., Lugato E., Meusburger K., Montanarella L., Alewell C. (2015). The new assessment of soil loss by water erosion in Europe. Environmental Science & Policy 54:438-447. https://doi.org/10.1016/j.envsci.2015.08.012

Panagos P., Standardi G., Borrelli P., Lugato E., Montanarella L., Bosello F. (2018). Cost of agricultural productivity loss due to soil erosion in the European Union:From direct cost evaluation approaches to the use of macroeconomic models. Land Degradation & Development 29: 471-484. https://doi.org/10.1002/ldr.2879

Pardini G., Gispert M., Dunjó G. (2004). Relative influence of wildfire on soil properties and erosion processes in different Mediterranean environments in NE Spain. Science of the Total Environment 328: 237–246. DOI:10.1016/j.scitotenv.2004.01.026.

Pasalodos-Tato M. & Pukkala T. (2007). Optimising the management of even-aged *Pinus sylvestri*s L. stands in Galicia, north-western Spain. Annals of Forest Science. 64, 787-798. DOI: 10.1051/forest:2007059

Pasalodos-Tato M., Pukkala T., Calama R., Cañellas I., Sánchez-González M. (2016). Optimal management of Pinus pinea stands when cone and timber production are considered. European Journal of Forest Research 135:607-619. DOI 10.1007/s10342-016-0958-7

Pausas J.G., Bladé C., Valdecantos A., Seva J.P., Fuentes D., Alloza J.A., Vilagrosa A., Bautista S, Cortina J., Vallejo R. (2004a). Pines and oaks in the restoration of Mediterranean landscapes of Spain: new perspective for an old practice—a review. Plant Ecology 171: 209–220. DOI:10.1023/B:VEGE.0000029381.63336.20.

Pausas J.G., Ribeiro E., Vallejo R. (2004b). Post-fire regeneration variability of *Pinus halepensis* in the eastern Iberian Peninsula. Forest Ecology and Management, 203:251-259. https://doi.org/10.1016/j.foreco.2004.07.061

Peura M., Triviño M., Mazziotta A., Podkopaev D., Juutinen A., Mönkkönen M. (2016). Managing boreal forests for the simultaneous production of collectable goods and timber revenues. Silva Fennica, 50:1672. 17 p. http://dx.doi.org/10.14214/sf.1672

Pintaldi E., Hudek C., Stanchi S., Spiegelberger T., Rivella E., Freppaz M. (2017). Sustainable soil management in ski areas: Threats and challenges. Sustainability 9:2150. doi:10.3390/su9112150

Poesen J., Vandaele K., Van Wesemael B. (1996). Contribution of gully erosion to sediment production on cultivated lands and rangelands. Erosion and sediment yield: Global and regional perspectives, proceedings of the Exeter symposium, July 1996. IAHS Publ. no. 236.

Poesen J., Nachtergaele J., Verstraeten G., Valentin C. (2003). Gully erosion and environmental change: importance and research needs. Catena 50:91–133. DOI:10.1016/S0341-8162(02)00143-1.

Poyatos R., Latron J., Llorens P. (2003). Land use and land cover change after agricultural abandonment: the case of a Mediterranean mountain area Catalan Pre-Pyrenees. Mountain Research and Development 23: 362–368.
https://doi.org/10.1659/0276-4741(2003)023[0362:LUALCC]2.0.CO;2

Provost F. & Domingos P. (2003). Tree induction for probability based ranking. Mach Learn 52(3):199–215

Pukkala T. (2002). Measuring non wood forest outputs in numerical forest planning. In: Pukkala, T. (ed.) Multi – objective forest planning, Kluwer Academic Publishers, Netherlands, pp. 173 – 207.

Rautiainen O., Pukkala T., Miina J. (2000). Optimising the management of even –aged Shorea robusta stands in southern Nepal using individual tree growth models. Forest Ecology and Management. 126, 417-429.

Razafindrabe B.H.N., He B., Inoue S., Ezaki T., Shaw R. (2010). The role of stand density in controlling soil erosion: implication to sediment – related disasters in Japan. Environ Monit Assess. 160: 337–354. DOI: 10.1007/s10661-008-0699-2

Renard K.G., Foster G.R., Weessies G.A., McCool D.K., Yoder D.C. (1997). Predicting soil erosion by water: a guide to conservation planning with the Revised Universal Soil Loss Equation (RUSLE). US Department of Agriculture, Agriculture Handbook 703.

Roces-Díaz J.V., Vayreda J., Banqué-Casanovas M., Cusó M., Anton M., Bonet J.A., Brotons L., De Cáceres M., Herrando S., Martínez de Aragón J., de-Miguel S., Martínez-Vilalta J. (2018). Assessing the distribution of forest ecosystem services in a highly populated Mediterranean region. Ecological Indicators 93: 986-997. https://doi.org/10.1016/j.ecolind.2018.05.076

Rodríguez J.P., Beard T.D. Jr, Bennett E.M., Cumming G.S., Cork S.J., Agard J., Dobson A.P., Peterson G.D. (2006). Trade-offs across space, time, and ecosystem services. Ecology and Society. 11, 28.
http://www.ecologyandsociety.org/vol11/iss1/art28/

Romero-Diaz A., Cammeraat L.H., Vacca A., Kosmas C. (1999). Soil erosion at three experimental sites in the Mediterranean. Earth Surface Processes and Landforms 24: 1243–1256.
https://doi.org/10.1002/(SICI)1096-9837(199912)24:13<1243::AID-ESP43>3.0.CO;2-Z

Scarascia-Mugnozza G., Oswald H., Piussi P., Radoglou K. (2000). Forests of the Mediterranean region: Gaps in knowledge and research needs. For. Ecol. Manag. 132: 97–109. https://doi.org/10.1016/S0378-1127(00)00383-2

Shakesby R.A. (2011). Post-wildfire soil erosion in the Mediterranean: review and future research directions. Earth-Science Reviews 105:71-100.
doi:10.1016/j.earscirev.2011.01.001

Selkimäki M. & Mola-Yudego B. (2011). Estimating and modelling the resistance of nature to path erosion in Koli National Park, Finland. Boreal Environment Research 16: 218-228.

Serpa D., Nunes J.P., Santos J., Sampaio E., Jacinto R., Veiga S., Lima J.C., Moreira M., Corte-Real J., Keizer J.J., Abrantes N. (2015). Impacts of climate and land use changes on the hydrological and erosion processes of two contrasting Mediterranean catchments. Science of the Total Environment 538: 64-77. http://dx.doi.org/10.1016/j.scitotenv.2015.08.033

Therneau T.M. & Atkinson E.J. (1997). An introduction to recursive partitioning using the RPART routines. Technical Report 61, Mayo Clinic, Section of Statistics.

Thomas A.D., Walsh R.P.D., Shakesby R.A. (1999) Nutrient losses in eroded sediment after fire in eucalyptus and pine forest in the Mediterranean environment of Northern Portugal. Catena 36: 283-302.

Thornthwaite C.W. (1948). An approach toward a rational classification of climate. Geographical Review 38:55–94. DOI: 10.2307/210739

Tóth N. & Pataki B. (2007). On classification confidence and ranking using decision trees. In: Proceedings of 11th international conference on intelligent engineering systems, pp 133–138.

Trasobares A. & Pukkala, T. (2004). Optimising the management of uneven-aged Pinus sylvestris L. and Pinus nigra Arn. Mixed stands in Catalonia north-east Spain. Annals of Forest Science. 61, 747-758. DOI: 10.1051/forest:2004071

Vandekerckhove L., Poesen J., Oostwoudwijdenes D., Nachtergaele J., Kosmas C., Roxo M.J., de Figueiredo T. (2000). Threshold for gully initiation and sedimentation in Mediterranean Europe. Earth Surface Processes and Landsforms 25: 1201–1220. DOI:10.1002/1096-9837(200010)25:11<1201::AID-ESP131>3.0.CO;2-L.

Van Liedekerke M., Jones A., Panagos P. (2006). ESDBv2 Raster Library a set of rasters derived from the European Soil Database distribution v2. 0. European Commission and the European Soil Bureau Network, CDROM, EUR, 19945.

Valinger E. & Fridman J. (1999). Models to assess the risk of snow and wind damage in pine, spruce, and birch forest in Sweden. Environmental Management 24:209-217.

Vanwalleghem T., Van Den Eeckhaut M., Poesen J., Govers G., Deckers J. (2008). Spatial analysis of factors controlling the presence of closed depressions and gullies under forest: application of rare event logistic regression. Geomorphology 95: 504–517. DOI:10.1016/j.geomorph.2007.07.003.

Vélez R. (1986). Fire prevention in Aleppo pine forests. Prévention d' incendies das les forêts de pind' Alep. *Le pin d'Alep et le pin brutia dans la sylviculture méditerranéenne.* Paris : CIHEAM, 1986:67-178 (Options Méditerranéennes: Série Etudes; n.1986-I)

Wade C.R., Bolding M.C., Aust W.M., Lakel W.A. III (2012). Comparison of five erosion control techniques for bladed skid trails in Virginia. Southern Journal of Applied Forestry 36: 191-197. https://doi.org/10.5849/sjaf.11-014

Wischmeier W.H., Smith D.P. (1978). Predicting rainfall erosion losses–a guide for selection for conservation planning. Agricultural Handbook (US Dept of Agriculture) 537.